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Instituto de Ciências Biológicas
Departamento de Genética, Ecologia e Evolução
Programa de Pós-Graduação em Ecologia, Conservação e Manejo da Vida Silvestre

Karoline Hellen Madureira de Melo

AVALIAÇÃO DE INTEGRIDADE ECOLÓGICA DE RIACHOS URBANOS REABILITADOS

Belo Horizonte

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Karoline Hellen Madureira de Melo

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Versão final

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Orientador: Prof. Dr. Marcos Callisto
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No dia 19 de junho de 2023, às 14:00 horas, sala 163, bloco K3, teve lugar a defesa de dissertação de mestrado no Programa de Pós-Graduação em Ecologia, Conservação e Manejo da Vida Silvestre, de autoria do(a) mestrando(a) Karoline Hellen Madureira de Melo, orientando(a) do Professor Marcos Callisto, intitulada: **"Avaliação de integridade ecológica de riachos urbanos reabilitados".** Abrindo a sessão, o(a) Presidente da Comissão, Doutor(a) Marcos Callisto de Faria Pereira, após dar a conhecer aos presentes o teor das normas regulamentares do trabalho final, passou a palavra para o(a) candidato(a) para apresentação de seu trabalho. Estiveram presentes a Banca Examinadora composta pelos Doutores: Marcelo da Silva Moretti (UVV), Ricardo Ribeiro de Castro Solar (UFMG) e demais convidados. Seguiu-se a arguição pelos examinadores, com a respectiva defesa do(a) candidato(a). Após a arguição, apenas os senhores examinadores permaneceram no recinto para avaliação e deliberação acerca do resultado final, sendo a decisão da banca pela:

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Nada mais havendo a tratar, o Presidente da Comissão encerrou a reunião e lavrou a presente ata, que será assinada por todos os membros participantes da Comissão Examinadora.

Belo Horizonte, 19 de junho de 2023.

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À minha mãe, que, mais do que me dar a vida,
me ensinou a viver.

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“A alegria não chega no encontro do achado, mas faz parte do processo de busca. E ensinar e aprender não pode dar-se fora da procura, fora da boniteza e da alegria.”

Paulo Freire

Resumo

Os riachos urbanos são afetados por uma complexa combinação de estressores (p. ex., deposição de lixo, poluição das águas, canalização), que modificam o habitat físico, características da água, a composição de comunidades biológicas e processos ecológicos, alterando assim a estrutura e o funcionamento dos ecossistemas. Projetos de reabilitação têm sido desenvolvidos em diversos países com o objetivo de recuperar riachos urbanos. No entanto, este ainda é um assunto incipiente em regiões tropicais. A maioria dos estudos aborda aspectos estruturais, como qualidade de água, controle de sedimentos e eventos de inundação, sem considerar indicadores de funcionamento de ecossistemas. Aqui, avaliamos a estrutura e o funcionamento de riachos urbanos reabilitados há 15 anos em comparação a riachos nas melhores condições ecológicas disponíveis (referência), riachos com moderada alteração de habitat (consolidados) e riachos severamente degradados. Encontramos maior diversidade de habitat e integridade de zona ripária, riqueza de taxa sensíveis de macroinvertebrados e pontuações de índices bióticos (multimétricos) de macroinvertebrados nos riachos reabilitados em comparação aos degradados. Além disso, encontramos menor produção primária, concentrações de nutrientes, deposição de sedimentos e assoreamento em riachos reabilitados em comparação a riachos degradados. No entanto, comparando os riachos reabilitados com os consolidados, encontramos maior produção primária nos reabilitados. Também encontramos menor diversidade de habitat, riqueza de macroinvertebrados sensíveis e índices de macroinvertebrados e maior produção primária nos riachos reabilitados em comparação aos riachos em condições de referência. Esses resultados evidenciam que os riachos reabilitados estão em melhores condições estruturais e funcionais do que os degradados, mas não diferem significativamente dos consolidados, nem atingiram uma condição semelhante aos de referência. Concluímos que a reabilitação é eficaz em minorar o estado de degradação, evidenciado pela estrutura e funcionamento do ecossistema. Além disso, o uso combinado de indicadores funcionais e estruturais permitiu uma avaliação abrangente da integridade ecológica dos riachos e permitiu a detecção de diferenças entre categorias de riachos urbanos além daquelas detectadas com base em indicadores estruturais de qualidade de água. A reabilitação de riachos urbanos é uma perspectiva de melhoria de qualidade ecológica na América do Sul e, potencialmente, poderá inspirar outros países no hemisfério sul a seguir o exemplo que permitirá recuperar alguns dos serviços de ecossistema que os riachos urbanos reabilitados podem prestar às populações urbanas.

Palavras-chave: Decomposição. Indicadores estruturais. Indicadores funcionais. Integridade ecológica. Macroinvertebrados. Parque urbano. Processos ecológicos. Produção primária.

Abstract

Urban streams are affected by a complex combination of stressors (e.g., garbage disposal, water pollution, canalization), which modify the physical habitat, water characteristics, the composition of biological communities and ecological processes, thus altering the structure and functioning of ecosystems. Rehabilitation projects have been developed in several countries with the objective of recovering urban streams. However, this is still an incipient subject in tropical regions. Most studies address structural aspects, such as water quality, sediment control and flood events, without considering indicators of ecosystem functioning. Here, we evaluate the structure and functioning of urban streams rehabilitated 15 years ago in comparison to streams in the best available ecological conditions (reference), streams with moderate habitat alteration (consolidated), and severely degraded streams. We found higher habitat diversity and riparian zone integrity, richness of sensitive macroinvertebrate taxa, and macroinvertebrate biotic index (multimetric) scores in the rehabilitated compared to the degraded streams. Additionally, we found lower primary production, nutrient concentrations, sediment deposition, and siltation in rehabilitated streams compared to degraded streams. However, comparing the rehabilitated streams with the consolidated ones, we found a higher primary production in the rehabilitated streams. We also found lower habitat diversity, richness of sensitive macroinvertebrates and macroinvertebrate indices, and higher primary production in rehabilitated streams compared to streams under reference conditions. These results show that the rehabilitated streams are in better structural and functional conditions than the degraded ones, but they do not differ significantly from the consolidated ones, nor did they reach a condition similar to the reference ones. We conclude that rehabilitation is effective in reducing the state of degradation, evidenced by the structure and functioning of the ecosystem. Furthermore, the combined use of functional and structural indicators allowed a comprehensive assessment of the ecological integrity of streams and allowed the detection of differences among categories of urban streams beyond those detected based on structural indicators of water quality. The rehabilitation of urban streams is a prospect for improving ecological quality in South America and, potentially, could inspire other countries in the southern hemisphere to follow the example that will allow recovering some of the ecosystem services that rehabilitated urban streams can provide to urban populations.

Key-words: Decomposition. Structural indicators. Functional indicators, Ecological integrity. Macroinvertebrates. Urban Park. Ecological processes. Primary production.

SUMÁRIO

INTRODUÇÃO

Ecossistemas de água doce prestam importantes serviços ecossistêmicos às populações humanas, incluindo geração de energia elétrica, abastecimento de água e irrigação de culturas (Hunter et al., 2019). Além disso, hospedam elevada riqueza e abundância de espécies, grande parte endêmica e sensível a alterações ambientais (Strayer & Dudgeon, 2010; Lepczyk et al., 2017). De acordo com a Agencia Nacional de Águas (Brasil, 2005), o Brasil está entre os países com a maior reserva de água doce no mundo, cerca de 12% do volume total disponível. Contudo, apesar de sua importância, ecossistemas aquáticos continentais são frequentemente ameaçados pela alta pressão humana, que resulta, por exemplo, na impermeabilização de solos, alteração de regime de vazão e eutrofização (Violin et al., 2011; Elosegi & Sabater, 2013).

Nas últimas décadas a expansão urbana tem sido um dos principais fatores de redução de qualidade de água em todo o mundo (Booth et al., 2016). Em geral, muitas cidades instalam-se e desenvolvem-se em áreas com disponibilidade de recursos hídricos (Wantzen et al., 2016). Atualmente, cerca de 55% da população mundial concentra-se em áreas urbanas e a expectativa é que chegue a 70% até 2050 (FAO, 2019; ONU News, 2019). A conjuntura econômica que concentra melhores oportunidades de emprego e desenvolvimento nos centros urbanos é uma das responsáveis por esse fenômeno. Neste contexto, riachos urbanos são cada vez mais impactados pelo desenvolvimento de cidades, muitas vezes desordenado e insustentável ambientalmente, levando a perdas de qualidade ambiental, com alteração a sua estrutura e funcionamento (Figura 1) (Konrad & Booth, 2005; Pompeu et al., 2005; Booth et al., 2016). Dentre os diferentes impactos, destacam-se a poluição de água, aumento no aporte de nutrientes, alterações no regime de vazão, modificações nos canais de riachos e zonas ripárias, acelerada perda de biodiversidade e aumento de invasões por espécies não nativas (Paul & Meyer, 2008; Carvalho-Santos et al., 2016; Wantzen et al., 2019).

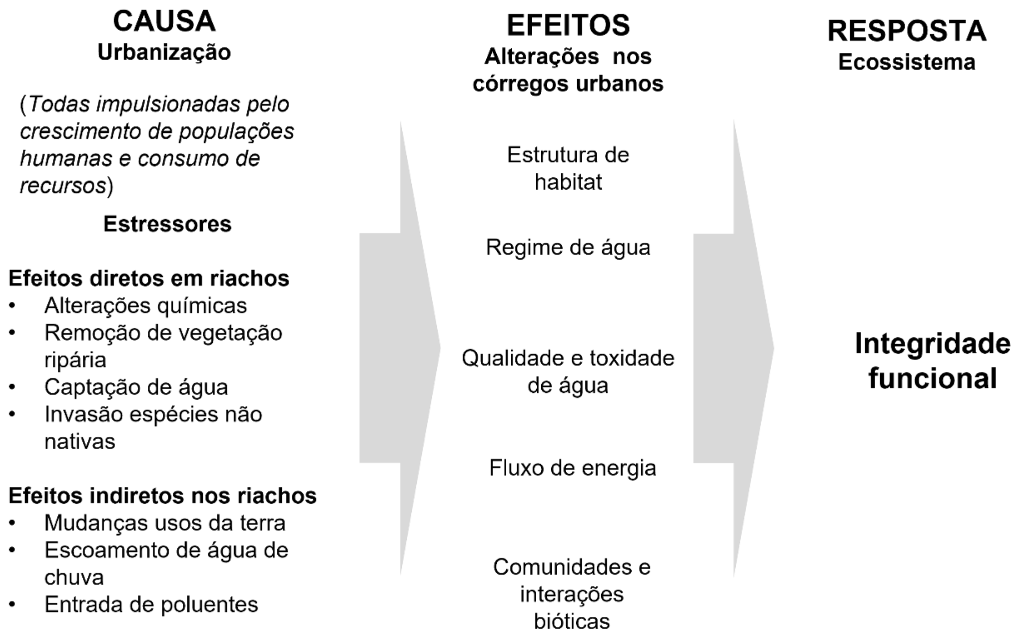


Figura 1. *Modelo conceitual de diferentes estressores resultantes de ações humanas que alteram a condição ecológica de riachos (adaptado de Booth et al., 2004).*

Quando em boas condições ecológicas, riachos urbanos atuam como importantes reservatório de biodiversidade local e regional, fonte de bens e serviços às populações ribeirinhas (Hunter et al., 2019; Feio et al., 2021; Ferreira et al., 2023; Xiao et al., 2023). Nas cidades, é possível aliar desenvolvimento urbano à conservação de riachos através de ações de conservação de cursos d'água, proteção de nascentes e recuperação das zonas ripárias (van Den Brandeler et al., 2019). Essas medidas podem, também, ser aplicadas na reabilitação de riachos impactados por atividades antrópicas, oferecendo condições para a melhoria de riachos degradados (Wantzen et al., 2019, Macedo & Magalhães Jr., 2020, Feio et al., 2021), trazendo ganhos sociais e culturais, valorização imobiliária e melhorias nos indicadores de saúde humana (Golgher et al., 2023).

Em diferentes partes do mundo o monitoramento de qualidade de água tem avaliado a extensão de danos causados por atividades antrópicas e subsidiado o planejamento de ações de reabilitação de bacias hidrográficas (Reid et al., 2019; Wantzen et al., 2019). Entretanto, a reabilitação de riachos é um tema ainda pouco estudado em regiões tropicais (Wantzen et al., 2019; Feio et al., 2021). No Brasil, os estudos ecológicos em riachos urbanos são recentes e carecem de mais informações sobre as condições ecológicas de bacias hidrográficas impactadas, conservadas e reabilitadas, bem como de investimentos públicos para melhoria de qualidade ambiental (Sá Costa et al., 2010; Macedo et al., 2011). Há, portanto, lacunas de conhecimento sobre como está a integridade ecológica em riachos urbanos tropicais. É estratégico e urgente o desenvolvimento de pesquisas ecológicas para a melhor compreensão da integridade ecológica de riachos urbanos tropicais e subsidiar a gestão e formulação de estratégias de conservação. Através da compreensão do estado de integridade funcional de riachos urbanos é possível uma melhor gestão e formulação de estratégias de conservação e reabilitação para esses recursos hídricos (Gessner & Chauvet, 2002; Barquín & Scarsbrook, 2008).

Em Belo Horizonte, a 3ª maior capital brasileira, as discussões relacionadas à conservação de riachos e drenagem urbana ganharam visibilidade a partir da década de 1970, devido à intensificação de problemas ambientas causados pela urbanização (Macedo & Magalhães Jr., 2020). Desde então, alguns planos têm sido implementados na cidade, como o "Plano metropolitano de águas pluviais e proteção contra cheias da RMBH" de 1975 e o "Plano de urbanização e saneamento de Belo Horizonte – PLANURBS" de 1979 (Duarte, 2009). No ano de 2003, a prefeitura elaborou o "Programa de Recuperação Ambiental e Saneamento dos Fundos de Vale e dos Córregos em Leito Natural de Belo Horizonte", conhecido como DRENURBS, que se tornou um dos melhores exemplos de programas de recuperação de cursos d'água no Brasil (PBH, 2010; Duarte, 2009; Macedo & Magalhães Jr., 2020; Macedo et al., 2022).

O DRENURBS teve como objetivo a despoluição de cursos d'água, o controle de produção de sedimentos e a integração de recursos hídricos no cenário urbano através de ações de tratamento, reabilitação de cursos d'água e implantação de parques ecológicos (PBH, 2003; 2010). Entre os anos de 2006 e 2008, o programa construiu três parques lineares em áreas com histórico de ocupação e pressões antrópicas (PBH, 2010; Macedo & Magalhães Jr., 2020). Estudos realizados por Macedo et al. (2022) indicaram melhorias significativas na qualidade de água comparando com períodos anteriores à reabilitação. Contudo, não existem informações referentes à capacidade de manutenção de processos ecológicos nestes riachos urbanos nem a sua integridade ecológica. Esta dissertação vem, portanto, preencher esta lacuna de conhecimento ao levantar informações sobre a integridade ecológica dos riachos, com uma abordagem que combina indicadores estruturais e funcionais de qualidade ambiental.

Abordagens de estudos baseadas em indicadores estruturais buscam avaliar a composição química, física e microbiológica da água e organismos bioindicadores (macroinvertebrados bentônicos, peixes, fitoplâncton, zooplâncton) (Wantzen et al., 2019). Estas abordagens são amplamente utilizadas, atendem aos usos de água no consumo doméstico, indústria e agricultura, mas não fornecem todas as respostas sobre a condição ambiental e integridade ecológica desses ecossistemas (Silveira, 2004). Portanto, a utilização apenas de indicadores estruturais pode não incluir informações relevantes para a avaliação completa da condição ecológica de riachos (Niyogi et al., 2013; Feckler & Bundschuh, 2020). Assim, idealmente deve-se combinar o uso de indicadores estruturais e indicadores funcionais, para possibilitar uma visão holística da integridade ecológica de riachos (Jørgensen, 2007; Friberg, 2014).

Apesar de pouco utilizados em programas de biomonitoramento (Ferreira et al., 2020; Feio et al., 2021), processos ecológicos oferecem informações acerca da integridade funcional de ecossistemas de água doce e, assim, complementam os indicadores estruturais e permitem uma avaliação integrada de qualidade ambiental em bacias hidrográficas urbanas (Dangles et al., 2004; Von Bertrab et al., 2013; Ferreira et al., 2020). Os processos ecológicos refletem os fluxos de energia e matéria no ecossistema, com influência do efeito legado (Schiller et al., 2008; Pereda et al, 2019). Por outro lado, características físicas e químicas são eficientes para a avaliação de condição momentânea, como uma fotografia da condição atual (Callisto et al., 2019). Dentre os processos ecológicos, a produção primária, a decomposição de detritos vegetais, o transporte e a deposição de sedimentos e macrofauna bentônica são importantes processos a serem considerados para preencher essas lacunas de conhecimento sobre o funcionamento de riachos urbanos (Feio et al., 2010; Ranta et al., 2021).

Os ecossistemas aquáticos que recebem aporte excessivo de nutrientes (fósforo e nitrogênio) têm sua produção primária aumentada. A eutrofização das águas pode ser utilizada na identificação de tipos de estressores e qualidade ecológica de riachos (Marques et al., 2003; Ansari et al., 2012; Kiffer et al., 2018). O processo de decomposição de detritos foliares de zonas ripárias é uma métrica das condições de riachos e tipos de estressores (Feio et al., 2010; Ferreira et al., 2022). Em geral, este processo é estimulado por enriquecimento moderado de nutrientes (Woodward et al., 2012; Rosemond et al., 2015), conservação da zona ripária (Casoti et al., 2015), porém inibido por acidificação das águas (Ferreira & Guérold, 2017) e perda de decompositores (Boyero et al., 2015).

O transporte e a deposição de sedimentos podem ser utilizados como indicadores de assoreamento de riachos e estado de conservação de suas margens (Von Bertrab et al., 2013; dos Reis Oliveira et al., 2018). A entrada de sedimentos finos em riachos deve-se principalmente à erosão das margens e é afetada pelo uso da terra na microbacia e perda de vegetação ciliar (Allan, 2004). O aumento no aporte de sedimentos pode indicar necessidade de recuperação de suas margens. O assoreamento de leitos de riachos pode dificultar a mobilidade de organismos aquáticos (Jones et al., 2012; Wood et al., 2016), alterar a disponibilidade de habitats (Burton & Johnston, 2010), afetar a disponibilidade de fontes alimentares (Schofield et al., 2004) e facilitar o escoamento de fertilizantes e pesticidas de áreas agrícolas para a água (Allan, 2004; dos Reis Oliveira et al., 2018), alterando a estrutura e composição da fauna bentônica (dos Reis Oliveira et al., 2020).

Essa dissertação está em formato de manuscrito a ser submetido a uma revista científica internacional, e tem como objetivo avaliar processos ecológicos e inferir sobre a integridade ecológica de riachos urbanos reabilitados. A hipótese testada é de que a reabilitação de riachos urbanos reestabelece as características estruturais e os processos ecológicos e recupera a integridade ecológica desses ecossistemas aquáticos.

É prevista uma correlação positiva entre os indicadores de integridade ecológica e o nível de conservação ambiental de riachos urbanos (Figura 2-A), onde, os riachos em condições de referência (“least disturbed”) devem apresentar valores mais altos para os indicadores de integridade ecológica, seguidos por riachos consolidados (implantados há mais de 20 anos) e por riachos reabilitados, e por último riachos degradados. Espera-se que a taxa de decomposição de detritos foliares de zonas ripárias seja positivamente correlacionada com o grau de conservação ambiental (Figura 2-B), enquanto o transporte de sedimentos e produção primária sejam negativamente correlacionados (Figuras 2-C e D). Por fim, espera-se que os resultados encontrados para a estrutura e composição das assembleias de macroinvertebrados, qualidade física, química e bacteriológica da água indiquem melhora nos indicadores de qualidade ambiental dos riachos urbanos reabilitados, aproximando-os do estado atual de riachos consolidados e em condições de referência.

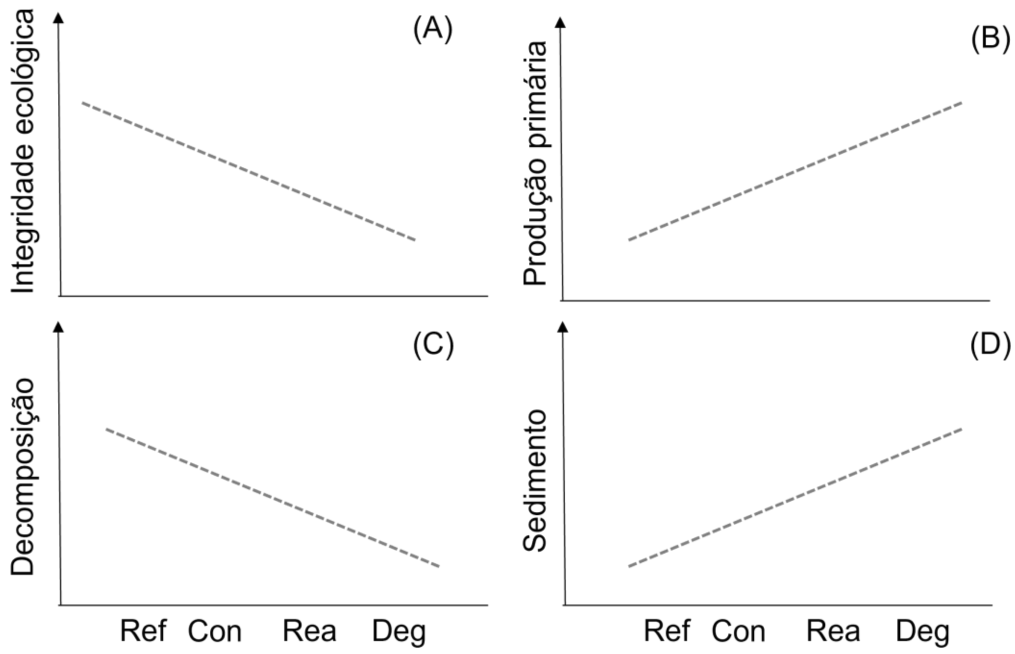


Figura 2. *Relação esperada entre os 4 tratamentos de riachos urbanos e A) integridade ecológica, B) taxa de decomposição de detritos foliares, C) produção primária e D) deposição de sedimentos. Ref = referência, Con = consolidado, Rea = reabilitado e Deg = degradado.*

Foram estudados riachos localizados em parques urbanos na região metropolitana de Belo Horizonte, comparando quatro cenários ambientais: i) riachos em condições de referência (“least disturbed”) (Figura 3-A a 3-C); ii) riachos em parques urbanos consolidados (que sofrem pressões urbanas, mas que foram minimamente perturbados ao longo do tempo) (Figura 3-D a 3-F); iii) riachos reabilitados em parques urbanos no "Programa DRENURBS” entre 2006-2008 (Figura 3-G a 3-I) e; iv) riachos degradados em área urbana (Figura 3-J a 3-L).

Os riachos de referência (“least disturbed”), que correspondem ao melhor estado ecológico possível em uma região para a qualidade da água, diversidade de habitat e integridade da zona ribeirinha, estão localizados no: (i) Parque Estadual da Serra do Rola Moça (20°03'39"S, 44°03'01"W), área de captação de água para consumo humano pela Companhia Estadual de Saneamento (Figura 3-A), (ii) Parque Municipal das Mangabeiras (19°56'43"S, 43°54'22"W), uma área de proteção ambiental com 59 nascentes e extensa vegetação nativa (Figura 3-B), e (iii) Parque Municipal Roberto Burle Marx (20°00 '02"S, 43°59'47"W), uma unidade de conservação urbana com extensa vegetação nativa (Figura 3-C).

Figura 3. *Riachos estudados. Riachos de referência: (A) Parque Estadual da Serra do Rola Moça, (B) Parque Municipal das Mangabeiras, e (C) Parque Municipal Roberto Burle Marx. Riachos consolidados: (D) Parque Municipal Aggeo Pio Sobrinho, (E) Parque Municipal Jacques Custeau, e (F) Parque Municipal Lagoa do Nado. Riachos reabilitados: (G) Parque Ecológico Primeiro de Maio, (H) Parque Municipal Nossa Senhora da Piedade, e (I) Parque Municipal José Lopes dos Reis Baleares. Riachos degradados: (J) Parque Tecnológico de Belo Horizonte, (K) Jardim Zoobotânico de Belo Horizonte, e (L) Parque Vilarinho.*

Os riachos consolidados estão inseridos em áreas residenciais urbanas, com grau moderado de alteração do habitat e que, historicamente, não foram afetados por fontes de degradação ambiental (por exemplo, contaminação de esgoto e lixo ou desmatamento das margens). Portanto, esses riachos se consolidaram dentro da área urbana nas condições encontradas atualmente sem a necessidade de intervenção humana como programas de reabilitação. Os córregos consolidados estão localizados no: (i) Parque Municipal Aggeo Pio Sobrinho (19°58'41"S, 43°58'17"W), que possui extensa vegetação natural e acesso restrito a visitantes (Figura 3-D), (ii) Parque Municipal Jacques Cousteau (19°58'20"S, 43°59'10"W), que possui uma extensa área de vegetação natural no local do córrego, mas que possui grandes depósitos de sedimentos finos no leito do córrego, provavelmente devido à erosão a montante (Figura 3-E), e (iii) Parque Municipal da Lagoa do Nado (19°50'28"S, 43°57'25"W), que possui mata ciliar estreita, marcas de erosão nas margens e grandes depósitos de areia (Figura 3-F).

Os riachos reabilitados, córregos fortemente degradados no passado, e sofreram intervenção para promover a reabilitação ecológica, estão localizados no: (i) Parque Ecológico Primeiro de Maio (19°51'16"S, 43°55'48"W) (Figura 3-G), (ii) Parque Municipal Nossa Senhora da Piedade (19°50’51’’S, 43° 55’38’’W) (Figura 3-H), e (iii) Parque Municipal José Lopes dos Reis Baleares (19°48'07"S, 43°57'54"W) (Figura 3-I).

Os riachos degradados, correspondentes aqueles ainda se encontram severamente degradados pelas atividades urbanas e com grande comprometimento da qualidade ambiental, estão localizados em: (i) Parque Tecnológico de Belo Horizonte (19°53'03"S, 43°58'37" W), área com recorrentes acidentes de derramamento de óleo diesel, depósito de lixo e esgoto no córrego e próximo a rodovias (< 5m) (Figura 3-J), (ii) Jardim Zoobotânico de Belo Horizonte (19°51'43"S, 44°01' 04"W), área com deposição de esgoto doméstico, monoculturas de cana-de-açúcar nas margens e canalização (Figura 3-K), e (iii) Parque Vilarinho (19°48'05"S, 43°59'2"W), onde há deposição de esgoto doméstico e lixo no córrego, ocupação humana das margens e várzeas e presença de animais domésticos no córrego como cavalos e porcos (Figura 3-L).

Para avaliação da integridade ecológica dos riachos, foram avaliados indicadores estruturais – caracterização física e química da água e do canal, diversidade de habitats e integridade da zona ripária, e estrutura e composição da comunidade de macroinvertebrados bentônicos – e indicadores funcionais – produção primária, decomposição de folhas, e deposição de sedimentos (Figura 4).

Figura 4. *Fotos dos trabalhos em campo e laboratório. A) Fixação dos tapetes e pedras para análise de deposição de sedimentos e de produção primária, respectivamente, B) medição in situ de parâmetros físicos e químicos da água, C) amostragem de macroinvertebrados bentônicos, D) substratos fixados em riacho, E) fixação de 'litter bags' de malha fina e malha grossa para avaliação de decomposição de detritos foliares, F) coleta dos substratos após 30 dias de incubação em riacho, G) identificação de macroinvertebrados, H) medição de macroinvertebrados sob lupa, e I) remoção de água sobrenadante por sucção para análise de sedimentos..*

MANUSCRITO

The way from degradation to reference conditions: rehabilitation of urban streams improves their structure and functioning

Karoline H. Madureira[a]*, Verónica Ferreira[b] & Marcos Callisto[a]

[a]Laboratório de Ecologia de Bentos, Departamento de Genética, Ecologia e Evolução, Instituto de Ciências Biológicas, Universidade Federal de Minas Gerais, Avenida Antônio Carlos, 31270-901 Belo Horizonte, Brazil

[b]MARE – Marine and Environmental Sciences Centre, ARNET – Aquatic Research Network, Department of Life Sciences, University of Coimbra, Calçada Martim de Freitas, 3000–456 Coimbra, Portugal

E-mail addresses: veronica@ci.uc.pt, mcallisto13@gmail.com

*Corresponding author: karolhmadureira@gmail.com

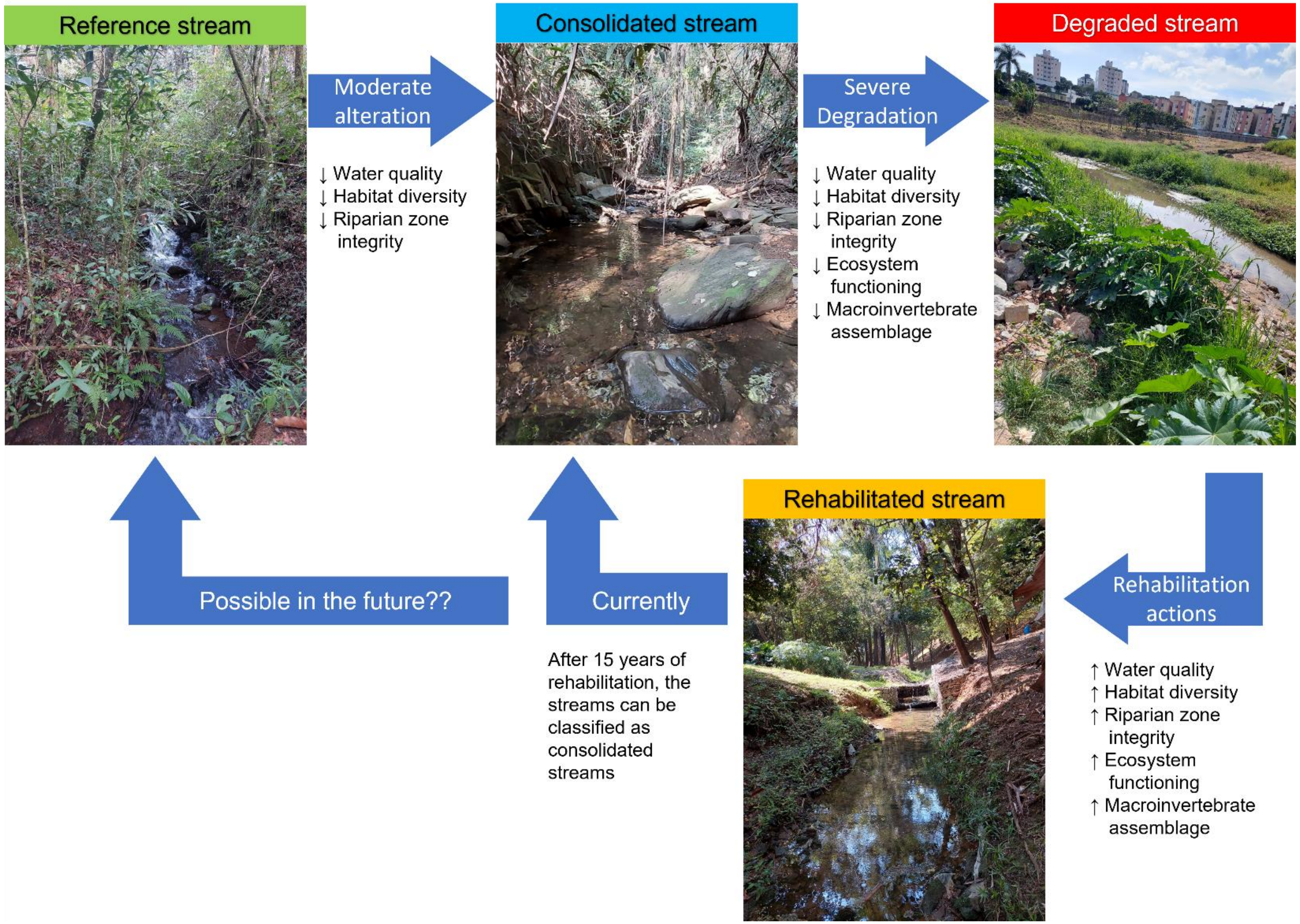
ORCID:
Karoline H. Madureira: 0000-0001-8651-1116
Verónica Ferreira: 0000-0001-7688-2626
Marcos Callisto: 0000-0003-2341-4700

Abstract

Urban streams are affected by a complex combination of stressors, which modify the physical habitat, water characteristics, composition of biological communities, and ecosystem processes, thus altering the structure and functioning of the ecosystem. Rehabilitation projects have been developed in several countries in order to recover urban streams. However, this is still a subject rarely reported in the tropical region. In addition, most studies focus on structural aspects, such as water quality, sediment control and flood events, without considering indicators of ecosystem functioning. Here, we evaluate the structure and functioning of 15-y old rehabilitated urban streams in comparison with streams in the best available ecological conditions (reference), streams with moderate habitat alteration (consolidated), and severely degraded streams. We found higher habitat diversity and riparian zone integrity, richness of sensitive macroinvertebrate taxa and scores of macroinvertebrate indices in the rehabilitated compared to the degraded streams. Additionally, we found lower primary production, nutrients concentrations, sediment deposition and siltation in rehabilitated compared to degraded streams. However, when comparing rehabilitated with consolidated streams, we found higher primary production in the rehabilitated streams. Also, we found lower habitat diversity, richness of sensitive macroinvertebrates and scores of macroinvertebrates indices and higher primary production in the rehabilitated compared to reference streams. These results indicate that rehabilitated streams are in better structural and functional condition than degraded streams, but do not strongly differ from streams in a moderately altered condition (consolidated), nor have they reached a condition similar to that of reference streams. We conclude that rehabilitation is effective in removing the stream from the degradation state, improving the structure and functioning of the ecosystem. Furthermore, the combined use of functional and structural indicators allowed for an integrative assessment of stream ecological integrity, and allowed detecting differences between stream categories beyond those detected based on single water quality indicators.

Key-words: Decomposition; ecological integrity; ecological process; functional indicator; macroinvertebrate; primary production; structural indicator; urban park.

Graphical Abstract



Introduction

Urban freshwater ecosystems provide important goods and services for human populations including regulatory services, such as mitigation of urban heat islands by regulating the microclimate and air humidity (Xiao et al., 2023), prevention of floods by increasing the water infiltration capacity (Yang & Zhang, 2011), and provision services such as the supply of good quality water (Hunter et al., 2019; Ferreira et al., 2023). Urban streams also constitute an important biodiversity reservoir by providing habitat and resources for many species, including birds, reptiles, mammals, insects, fishes, aquatic plants, and algae (Lepczyk et al., 2017).

However, urban streams are affected by a complex combination of human disturbances and commonly express degraded physical, chemical and biological conditions that have been collectively termed “urban stream syndrome” (Walsh et al., 2005; Hughes et al., 2014; Booth et al., 2016). To allow the growth of cities, natural characteristics of the environment are usually disregarded, forcing streams to fit into the urban design (Paul & Meyer, 2008). For instance, in order to constrain the streams and the natural flood pulses, drainage works are carried out with the channeling and linearization of the streams resulting in changes in the hydromorphology of the channel, removal of natural obstacles that could accumulate sediments and organic matter, and increases in water velocity (Paul & Meyer, 2008; Wantzen et al., 2019). Also, to increase space for human activities leads to the deforestation of riparian zones, causing increases in soil exposure and susceptibility to erosion, loss of shading, increases in water temperature, and decreases in plant organic matter inputs (Carvalho-Santos et al., 2016; Wantzen et al., 2019). In addition, the absence (total or partial) of sanitation and of efficient sewage and garbage collection, makes urban streams a common destination for waste disposal, resulting in water pollution and consequent increases in the concentrations of nutrients, heavy metals, and other contaminants (Paul & Meyer, 2008). These changes impact the structure of the stream ecosystem, altering the physical habitat (Wantzen et al., 2019), water characteristics (Bakure et al., 2020), and consequently the composition of biological communities that show, e.g., a decrease in biotic richness and sensitive species abundance (Birk et al., 2020; Firmiano et al., 2021), and an increase in non-native species (Gaertner et al., 2017).

Ecosystem functioning (e.g., primary and secondary productions, decomposition of organic matter, and stream metabolism) is also affected, with consequences on the energy flow and nutrient dynamics, through the food web and transport to downstream reaches (von Schiller et al., 2008; Elosegui & Sabater, 2013; Pereda et al., 2019). In streams, the decomposition of organic matter is directly affected by the entry of pollutants, modification of the riparian forest, and, inhibition of decomposers and detritivores (Yule et al., 2015; Ferreira et al., 2020; Tagliaferro et al., 2020). In addition, primary production is stimulated by increased nutrient and light availability, both of which are enhanced by environmental degradation in urban streams (Feio et al., 2010; Mamum & An, 2019; Vicent et al., 2022).

Stream rehabilitation interventions can be used to recover ecosystems, improve the ecological conditions of urban streams, and ensure the provision of goods and services to human populations (Wantzen et al., 2019; Feio et al., 2021). Rehabilitation interventions aim to improve streams status by bringing them closer to their pre-disturbance or to a reference condition (Brierley & Fryirs, 2000). In general, rehabilitation implies the recovery of structural elements such as the physical and chemical quality of the water, channel realignments, channel physical complexity, and stability of the banks (Hughes et al., 2014; Espinosa et al., 2016). Rehabilitation also includes measures to preserve biodiversity and the riparian zone, such as the recovery of riparian vegetation, creation of environmental preservation areas, protection of springs and of aquatic organisms (Feio et al., 2021). In addition, rehabilitation projects may include the creation of linear parks or urban green areas for human use as leisure, sports practices, recreational trails, and environmental education activities (Porfiriev et al., 2017; Hunter et al., 2019; Macedo et al., 2022). These rehabilitation measures make it possible to recover the original channel morphology, improve the quality of the water and the riparian zone integrity, increase the diversity of habitats, and promote biodiversity recovery (Wantzen et al., 2019; Feio et al., 2021). Consequently, rehabilitation projects can ensure the functioning of the ecosystem and its capacity to offer goods and services, although, in general, this is not the objective of the interventions (Feio et al., 2021; Ranta et al., 2021).

Stream rehabilitation projects have been developed in several countries (Feio et al., 2021). However, rehabilitation of urban streams is still a subject that has been little studied in tropical regions (Macedo et al., 2011; Wantzen et al., 2019; Feio et al., 2021). In addition, most studies evaluating urban stream rehabilitation efficacy focus on structural aspects such as water quality, sediment control, and flood events (e.g., Pedraza et al., 2008; da Cruz e Souza & Ríos-Touma, 2018; Macedo et al., 2022), without considering indicators of ecosystem functioning (Wantzen et al., 2019). However, as ecosystem's structure and function are not always related (McKie & Malmqvist, 2009; Feckler & Bundschuh, 2020), both attributes must be considered for a holistic assessment of stream ecological integrity (Feio et al., 2010; Ranta et al., 2021; Brosed et al., 2022). In fact, functional approaches including ecological processes such as primary production, organic matter decomposition, and ecosystem metabolism (Feio et al., 2010; Ferreira et al., 2020, 2021) can provide important information about ecosystems functional integrity. Together, structural and functional indicators contribute to an integrative ecosystem assessment (von Schiller et al., 2008; Pereda et al, 2019; Ranta et al., 2021).

This study assessed the effects of rehabilitation on the structure and functioning of urban streams in the city of Belo Horizonte (Minas Gerais, Brazil), by comparing rehabilitated streams with streams in the best ecological conditions available (reference), streams with moderate habitat alteration but which were not rehabilitated (consolidated), and severely degraded streams (degraded), in terms of structural and functional indicators. Stream rehabilitation interventions were carried out from 2006 until 2008, when linear parks were created and actions were implemented to improve the water quality. The three rehabilitated streams are located at the Primeiro de Maio Ecological Park, José Lopes dos Reis Baleares Municipal Park, and Nossa Senhora da Piedade Municipal Park. Rehabilitation efficacy in these streams was previously studied by Macedo et al. (2022), who compared stream physical, chemical and biological parameters before, during and 10 years after rehabilitation. However, stream ecosystem functioning was not evaluated. In this sense, this study aims to evaluate the ecological integrity of rehabilitated urban streams by answering the question: Is rehabilitating effective in improving the structural and functional indicators of a previously degraded stream? Our hypothesis is that the structural and functional integrity of rehabilitated streams are closer to all consolidated streams and farther away from degraded streams. As rehabilitation interventions improve the structure and functioning of urban streams, we anticipate that rehabilitated streams will assume an intermediate level of integrity on the stream alteration gradient (changed < rehabilitated = consolidated < baseline) across all parameters studied (table 1).

Table 1. *Mechanisms that support the prediction about the effects of rehabilitation of urban streams on structural and functional parameters.*

Parameter	Mechanism	Rationale	Reference
Physical and chemical water quality	Dissolved oxygen concentration and canopy cover will be higher and biochemical oxygen demand, total dissolved solids, total nitrogen and total phosphorus concentrations and turbidity will be lower on reference, consolidated, followed by rehabilitated, and lastly degraded streams.	Rehabilitation actions interrupt the entry of polluting agents into the stream (sewage, garbage) and promote the improvement in water quality indicators.	Macedo et al. (2022)
Habitat diversity	Habitat diversity will be higher on reference, consolidated, followed by rehabilitated, and lastly degraded streams.	The rehabilitation process recovers the riparian zone and the natural characteristics of the stream, increasing habitat diversity and complexity.	Macedo et al. (2022)
Macroinvertebrate assemblage structure and composition	Richness and diversity of total taxa, richness and abundance of sensitive taxa and biotic indices scores will be higher and abundance of total taxa, richness and abundance of resistant taxa will be lower on reference, consolidated, followed by rehabilitated, and lastly degraded streams.	Rehabilitation of streams minimizes the presence of anthropogenic stressors that would select organisms with traits that confer resistance to environmental degradation, increasing taxa richness and diversity and the presence of sensitive taxa.	Castro et al. (2018); Al-Zankana et al. (2020); Firmiano et al. (2021)
Primary production	Chlorophyll a concentration will be lower and autotrophic index will be higher on reference, consolidated, followed rehabilitated, and lastly degraded streams.	High primary production rate occurs mainly in streams with excessive nutrient input and lower canopy cover.	Feio et al. (2010)
Organic matter decomposition	Remaining organic matter mass and decomposition rates will be changed in rehabilitated and degraded streams compared to reference streams.	The improvement in organic matter decomposition efficiency is due to the decrease in sedimentation and increase in pH, oxygenation and habitat heterogeneity that promote the diversity and activity of decomposers.	Ferreira et al. (2021)

Sediment deposition	Sediment deposition will be less on reference, consolidated, followed by rehabilitated, and lastly degraded streams.	The entry of fine sediments into the streams is mainly due to the erosion of the banks, associated with the suppression of the riparian vegetation. In the rehabilitated streams, works were carried out to contain the banks and recover the riparian zone.	von Bertrab et al. (2013)

Methods

Study streams

This study was conducted in 12 stream sites located within parks in the metropolitan region of Belo Horizonte (Fig. 1, Table S1), the third largest Brazilian metropolis (ca. 5.5 million inhabitants), during the drought period (May to September 2022). The streams were classified in four categories (three on each), according to the environmental conditions of the park where they are located: reference, consolidated, rehabilitated, and degraded.

Rehabilitated streams, which are heavily degraded streams that underwent intervention to promote ecological rehabilitation, are located at: (i) Primeiro de Maio Ecological Park (19°51’16”S, 43°55’48”W), (ii) José Lopes dos Reis Baleares Municipal Park (19°48’07”S, 43°57’54”W), and (iii) Nossa Senhora da Piedade Municipal Park (19°50’51”S, 43°55’38”W) (Fig. 1). The intervention in the rehabilitated streams was carried out by the “DRENURBS Program” of the Municipality of Belo Horizonte, with financial support from the Inter-American Development Bank (BID, 2008), between 2006 and 2008. Before the rehabilitation interventions, there was human occupation of the stream banks and floodplains, untreated sewage discharges into natural channels, garbage dumping, absence of riparian vegetation, erosion on the banks and siltation on stream beds. Linear Parks were created and actions were implemented to improve the stream water quality (Macedo et al., 2022), including the creation of a sewage collection network, the construction of gabions to stabilize the banks, the construction of a retention basin to control floods, the removal of houses from stream banks, and the installation of trails and structures for recreational activities (BID, 2008; PBH, 2012). Rehabilitated streams have sparse riparian woody vegetation, with a predominance of grasses, anthropic alterations upstream of the study stream site (channeling and residential occupation), free and easy access for the visiting population, and local gentrification (Macedo et al., 2022; Golgher et al., 2023).The reference ("least disturbed") streams, corresponding to the best ecological state possible in a region for water quality, habitat diversity, and riparian zone integrity, closer to natural conditions, are located at: (i) Serra do Rola Moça State Park (20°03'39"S, 44°03'01"W), an area where water is collected for human consumption by the State Sanitation Company, (ii) Mangabeiras Municipal Park (19°56'43"S, 43°54'22"W), an environmental protection area with 59 springs and extensive native vegetation, and (iii) Roberto Burle Marx Municipal Park (20°00'02"S, 43°59'47"W), an urban conservation unit with extensive native vegetation (Fig. 1). While all reference parks have structures for visitation and leisure, the areas where the reference streams are located have restricted human access. The streams originate in conservation areas and flow directly into the park, without interference from urban areas upstream of the study stream sites.

The consolidated streams are inserted in urban residential areas, with a moderate degree of habitat alteration and which, historically, have not been affected by sources of environmental degradation (e.g., sewage and garbage contamination or deforestation of the banks). Therefore, these streams have consolidated within the urban area under the conditions currently found without requiring human intervention such as rehabilitation programs. The consolidated streams are located at: (i) Jacques Cousteau Municipal Park (19°58'20"S, 43°59'10"W), which has an extensive area of natural vegetation at the stream site, but which has large deposits of fine sediment in the stream bed, probably due to erosion upstream, (ii) Lagoa do Nado Municipal Park (19°50'28"S, 43°57'25"W), which has narrow riparian vegetation, erosion marks on the banks and large sand deposits, and (iii) Aggeo Pio Sobrinho Municipal Park (19°58'41"S, 43°58'17"W), which has extensive natural vegetation and restrict access to visitors (Fig. 1).

Degraded streams, corresponding to streams that are still severely degraded due to urban activities and have severely compromised environmental quality, are located at: (i) Belo Horizonte Technological Park (19°53'03"S, 43°58'37"W), an area with recurrent diesel oil spill accidents, garbage and sewage deposit in the stream and near to the roads (< 5m), (ii) Belo Horizonte Zoobotanic Garden (19°51'43.6"S, 44°01'04"W), an area with deposition of domestic sewage, sugar cane monocultures on the banks and canalization, and (iii) Vilarinho Park (19°48'05"S, 43°59'2"W), which has deposition of domestic sewage and garbage in the stream, human occupation of the stream banks and floodplains, and presence of domestic animals in the stream such as horses and pigs (Fig. 1).

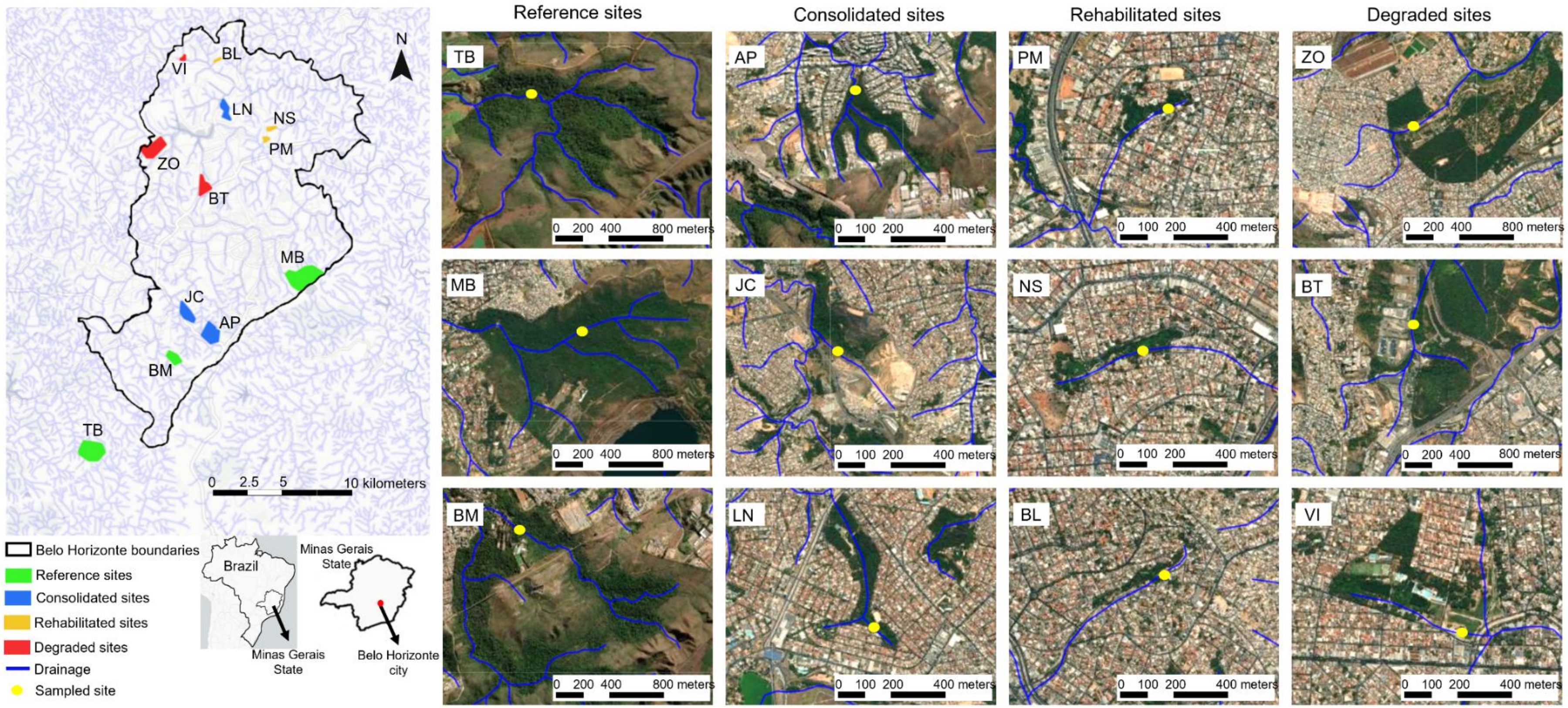


Figure 1. *Location of reference, consolidated, rehabilitated and degraded stream sites in Minas Gerais state, Brazil: RL = Serra do Rola Moça State Park, MB = Mangabeiras Municipal Park, BM = Roberto Burle Marx Municipal Park, JC = Jacques Cousteau Municipal Park, LN = Lagoa do Nado Municipal Park, AP= Aggeo Pio Sobrinho Municipal Park, PM = May Day Ecological Park, BL = José Lopes dos Reis Baleares Municipal Park, NS = Nossa Senhora da Piedade Municipal Park, BT = Belo Horizonte Technological Park, ZO = Belo Horizonte Zoobotanic Garden, VI = Vilarinho Park.*

Abiotic characterization

Physical and chemical water parameters were measured *in situ*, monthly for five months (May to September 2022), for each stream site, and included temperature (°C), dissolved oxygen concentration (mg/L) and saturation (%) (YSI ProSolo model), pH and oxi-redox potential (mV) (Digimed DM-2P), turbidity (NTU; Digimed DM-TU), electric conductivity (µS/cm), total dissolved solids (mg/L) and resistivity (KΩ/cm) (Digimed DM-3P). On the same occasions, 500 mL of water were collected in glass bottles for determination of the biochemical oxygen demand (BDO; mg/L) and 250 mL of water were collected in amber plastic bottles for determination of total phosphorus and total nitrogen concentrations (mg/L) (APHA, 2012).

Stream depth (m; average of three random measurements), wet width (m; average of three random measurements), flow velocity (m/s, average of three random measurements) and canopy cover (average of three random measurements, using free Canopy Capture® software) were measured on the same occasions, at each stream site.

The Habitat Diversity Rapid Assessment Protocol (Callisto et al., 2002) was applied in May 2022 to assess the physical condition of streams and their riparian areas. This protocol is based on the assessment of 22 visual environmental parameters related to the use and occupation of the riparian zone and apparent characteristics of the water (e.g., vegetation cover in the riparian zone, presence of anthropogenic changes, presence of erosion on the streambanks and silting up), scored from zero to four, and 11 visual environmental parameters related to flow conditions and substrate (e.g., types of background, sediment deposition, and water flow characteristics), scored from zero to five. The protocol is synthesized into a final score that reflects the conservation level of each site, where a score of 0–40 indicates degraded sites, 41–60 indicates altered sites, and >60 indicates reference sites (Callisto et al., 2002).

Benthic macroinvertebrates

We collected three benthic subsamples in each of 12 stream sites with a Surber sampler (30 cm aperture, 0.09 m^2 area, 0.250 mm mesh) in September 2022, preferably one subsample in pebbles/gravel, another in fine sediments (sand and silt), and a third in leaf deposits to capture the greatest biological diversity. The subsamples were individually placed in plastic bags and fixed with 70% ethanol. In the laboratory, the samples were washed over sieves (0.250 mm mesh) in running water, and organisms were sorted and identified to family level, except for the suborder Hydracarina, class Bivalvia and subclass Oligochaeta, under a magnifying glass using taxonomic keys (Merritt et al., 2008; Mugnai et al., 2010; Hamada et al., 2014; Hamada et al., 2018) (Table S2). Benthic macroinvertebrates were classified according to their tolerance to environmental degradation, as resistant, tolerant or sensitive, following Junqueira et al. (2000, 2018). The three subsamples for each stream were considered as one sample for the analyses.

Functional variables

Biofilm growth on artificial substrates

Biofilm growth rate, chlorophyll a (Chla) concentration and the autotrophic index were used as primary production indicators (APHA, 2012). Three clean slate stones (8 × 8 cm, 64 cm^2) were incubated in each stream for four consecutive 30-day periods (May to August 2022). After each incubation period (30 d), the stones were collected, enclosed individually in aluminum foil envelopes and taken to the laboratory, in a dark and refrigerated box.

The biofilm developing in the upper surface of each stone was removed with a cell scraper. The biofilm content in half of the scraped surface area (32 cm^2) was transferred to a pre-weighed porcelain crucible, oven-dried (60 °C, 48 h), weighed (± 0.01 mg) to determine dry mass (DM), incinerated (550 °C, 4 h), and weighed again to determine the ash mass. The biofilm ash-free dry mass (AFDM; mg) was estimated as the difference between DM and ash mass. The biofilm growth rate (Gr) was calculated as: Gr = Biofilm AFDM × a^{-1} × t^{-1}, where a is the scraped area of the stone (m^2) and t is the incubation time (days), and results were expressed as mg AFDM/m2/d.

The biofilm content in the second half of the scraped surface area (32 cm^2) was added to 100 mL of distilled water, vacuum filtered through a glass fiber filter, soaked in 10 mL acetone solution (90%) for ~ 20 hours in the fridge and macerated. The Chla concentration was determined spectrophotometrically (Thermo GENESYS™ 10UV UV-Vis) by reading the absorbance at 664, 665 and 750 nm (APHA, 2012), and results were expressed as mg Chla/m^2. The relative importance of autotrophs versus heterotrophs in the biofilm was calculated as the autotrophic index (AI): AI = Biofilm AFDM (mg) / Chla (mg).

Sediment deposition

Artificial grass mats (10 cm × 15 cm × 3 cm, 2 mm thick filaments and 42 filaments per cm^2) were used to estimate sediment deposition (von Bertrab et al., 2013). Three mats were fixed to the stream bed of each stream with clamps and iron rods, for four consecutive 30-day periods (May to September 2022). After each incubation period (30 d), the mats were carefully removed, enclosed individually in plastic bags with hermetic seals, and taken to the laboratory. In the laboratory, the mats were washed under running water, and the water and residues were collected in aluminum trays. The trays were left intact to allow the sediment to decant until the supernatant water became clear, which was later removed by suction, leaving only the wet sediment in the tray. The wet sediment was oven-dried (60 °C, 4 days) and weighed (± 0.01 mg) to determine the total sediment mass, and results were expressed as g.

Organic matter decomposition

Senescent leaves of *Bauhinia forficata* Link (Fabaceae) were collected at the Universidade Federal de Minas Gerais with nets installed below the canopies, in autumn (February – April 2022), and allowed to air dry at room temperature. *Bauhinia forficata* is a native, deciduous or semi-deciduous tree species, which is abundant in the study region (Lorenzi, 2002; Vaz et al., 2010). Because it is a pioneer fast-growing plant, it is commonly used in mixed plantations in ecological restoration projects of the riparian vegetation (Lorenzi, 2002, Vaz et al., 2010). Individuals of *B. forficata* were visually identified in 10 of the 12 parks studied, including the three rehabilitated parks.

About 3 g (± 0.01 mg) of air-dried leaves were placed in fine-mesh bags (FM; 17 × 10 cm, 0.5 mm mesh) to determine microbial-mediated leaf litter decomposition, and in coarse-mesh bags (CM; 10 × 14 cm, 1 cm mesh) to determine leaf litter decomposition mediated by both macroinvertebrates and microorganisms (i.e., total leaf litter decomposition) (Graça et al., 2005). Three leaf litter bags of each type (3 FM and 3 CM) were deployed at each stream on June 2022 and allowed to decompose for 60 days. After the incubation period, all litter bags were recovered, enclosed individually in plastic bags, and transported to the laboratory in a cooler. Leaf litter was removed from the mesh bags, carefully cleaned of sediments and associated organisms with running tap water on top of a sieve (0.5 mm mesh) to retain small leaf fragments, oven-dried (60 °C, 48 h), weighed (± 0.01 mg) to determine DM, incinerated (550 °C, 4 h), and weighed again to determine ash mass. Leaf litter AFDM remaining was estimated as the difference between DM and ash mass, and the percentage of AFDM remaining was estimated as: % AFDM remaining = AFDM remaining (g) / initial AFDM (g) × 100. The initial AFDM was estimated by multiplying the initial air-dried mass by a correction factor determined from an extra set of 12 litter bags, which were prepared as the experimental litter bags, but used on day 0 to estimate the initial AFDM (as described above).

Data analyses

As samples carried out over 5 months on each of 12 stream sites are pseudo-replicates, and there were no seasonal variations among months, the statistical analyses were performed on the average across the months for each stream, with the three streams within each stream category considered as replicates.

Physical and chemical water parameters, channel measures, habitat diversity and functional variables, except AFDM remaining, were compared among stream categories using one-way ANOVA, followed by Tukey's post-hoc tests to assess statistical differences among stream categories (reference, consolidated, rehabilitated and degraded). AFDM remaining were compared among streams categories, mesh size and their interaction using two-way ANOVA, followed by Tukey's post-hoc test.

The benthic macroinvertebrate communities were compared among stream categories regarding for different types of widely used indicators. (i) Assemblage structure indicators, including total taxa richness, total individual abundance, density of organisms per m^2, Shannon-Wiener diversity index, Simpson diversity index and Pielou evenness index. (ii) Assemblage composition indicators, including taxonomic composition, % EPT (Ephemeroptera, Plecoptera, Trichoptera) individuals, EPT/Chironomidae individuals ratio, EPT/(Chironomidae + Oligochaeta) individuals ratio, % Chironomidae individuals, % Chironomidae + Oligochaeta individuals, % sensitive individuals abundance, % tolerant individuals abundance, % resistant individuals abundance, % sensitive taxa richness, % tolerant taxa richness, and % resistant taxa richness. (iii) Macroinvertebrate biotic indices including Biological Monitoring Working Party (BMWP), Average Sore per Taxon (ASPT) following Junqueira et al. (2000, 2018), Benthic Multimetric Index (BMI) proposed by Ferreira et al. (2011), Macroinvertebrate Multimetric Index (MMI 1) proposed by Macedo et al. (2016), and MMI 2 proposed by Silva et al. (2017), for streams in Minas Gerais, Brazil.

Taxonomic composition was compared among stream categories by ANOSIM, followed by pairwise comparisons with Wilcoxon's test. Other macroinvertebrate indicators were compared among stream categories by one-way ANOVA, followed by Tukey's post-hoc when needed. Pearson's correlation index was calculated comparing indicators of macroinvertebrates and habitat diversity.

All statistical tests were performed using the software R (R Core Team 4.2.1). Data were transformed with log(x) whenever necessary as indicated in the statistics tables. Normality was assessed using the Shapiro-Wilk's test and homogeneity of variances using the Levene's test. Significance was established at $\alpha = 0.05$.

Results

Abiotic characterization

Stream categories differed in seven out of 16 physical and chemical parameters (one-way ANOVA, $p \leq 0.04$; Table S3, Table 2). Rehabilitated streams had lower biochemical oxygen demand than degraded streams, while they had higher conductivity and total dissolved solids, and lower resistivity and canopy cover than reference streams. Dissolved oxygen concentration and saturation, resistivity and canopy cover were higher, and conductivity, biochemical oxygen demand and total dissolved solids were lower in reference than the degraded streams (Table 2). Consolidated streams did not differ from reference streams, but had lower conductivity and total dissolved solids than degraded, and rehabilitated streams and lower biochemical oxygen demand than degraded streams (Table 2).

Habitat diversity scores significantly differed among stream categories (one-way ANOVA, $p < 0.01$; Table S3). Reference streams had the highest habitat diversity scores, followed by consolidated and rehabilitated streams that did not differ among each other, while degraded streams had the lowest scores (Fig. 2).

Table 2. *Physical and chemical variables of water quality in reference, consolidated, rehabilitated and degraded streams. Different letters indicate statistical differences among stream categories (one-way ANOVA followed by Tukey's test, p < 0.05).*

Metrics	Reference	Consolidated	Rehabilitated	Degraded	p-value
Electric conductivity (µS/cm)	**36.2 ± 30.7 a**	**119.5 ± 79.8 a**	**338.0 ± 62.6 b**	**318.6 ± 49.8 b**	**< 0.01**
Dissolved oxygen concentration (mg/L)	**8.4 ± 0.1 a**	**7.4 ± 1.0 ab**	**7.2 ± 0.4 ab**	**5.0 ± 2.1 b**	**0.04**
Dissolved oxygen saturation (%)	**101.5 ± 3.0 a**	**88.9 ± 10.3 ab**	**87.6 ± 2.2 ab**	**61.0 ± 25.1 b**	**0.04**
Biochemical oxygen demand (mg/L)	**0.5 ± 0.1 a**	**1.0 ± 0.4 a**	**1.2 ± 0.6 a**	**3.1 ± 0.8 b**	**< 0.01**
pH	7.67 ± 0.30 a	7.57 ± 0.08 a	7.79 ± 0.35 a	7.68 ± 0.30 a	0.81
Oxi-redox potential (mV)	153.4 ± 19.4 a	114.1 ± 9.8 a	138.2 ± 29.8 a	130.4 ± 15.9 a	0.19
Resistivity (KΩ/cm)[1]	**40.9 ± 37.9 a**	**10.5 ± 3.1 ab**	**3.8 ± 1.3 b**	**3.7 ± 0.9 b**	**< 0.01**
Water temperature (°C)	18.9 ± 1.4 a	19.2 ± 1.7 a	21.1 ± 1.9 a	20.3 ± 0.7 a	0.31
Total dissolved solids (mg/L)	**24.1 ± 18.4 a**	**72.6 ± 43.0 a**	**210.4 ± 55.8 b**	**191.4 ± 29.9 b**	**< 0.01**
Total nitrogen (mg/L)	6.97 ± 2.49 a	8.44 ± 3.26 a	7.97 ± 3.22 a	12.35 ± 3.94 a	0.87
Total phosphorus (mg/L)	0.103 ± 0.055 a	0.056 ± 0.062 a	0.069 ± 0.090 a	0.051 ± 0.046 a	0.65
Turbidity	1.0 ± 0.9 a	3.3 ± 2.2 a	3.3 ± 3.3 a	8.7 ± 10.5 a	0.43
Canopy cover (%)	**83.7 ± 2.6 a**	**78.6 ± 0.4 ab**	**71.6 ± 4.3 b**	**48.2 ± 41.9 b**	**0.01**
Depth (m)	0.096 ± 0.043 a	0.103 ± 0.067 a	0.124 ± 0.041 a	0.075 ± 0.025 a	0.66
Flow velocity (m/s)	**0.32 ± 0.22 a**	**0.04 ± 0.03 a**	**0.06 ± 0.05 a**	**0.63 ± 0.45 a**	**0.06**
Wet width (m)	2.54 ± 1.02 a	3.32 ± 0.67a	2.06 ± 0.25 a	2.51 ± 0.32 a	0.20

[1]Log(x)-transformed for the analysis

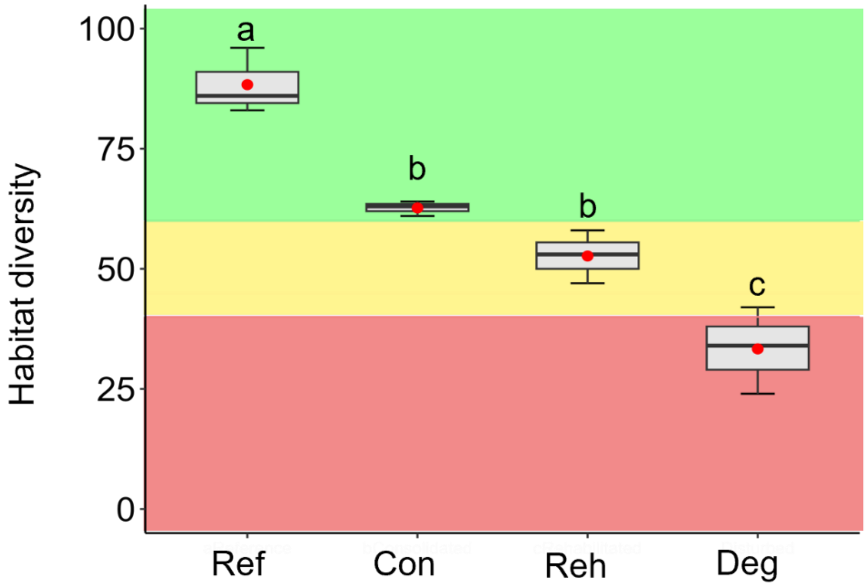


Figure 2. *Final scores of the Habitat Diversity Rapid Assessment Protocol (Callisto et al., 2002). Values < 40 indicate degraded streams (red band), values between 41 and 60 indicate altered streams (yellow band), and values > 60 indicate streams in reference conditions (green band). In the boxplot, the central line represents the median, the rectangle represents the 50% confidence interval, the vertical bars represent the data dispersion, and the red dot represents the mean. Different letters indicate statistical differences among stream categories (one-way ANOVA followed by Tukey's test, p < 0.05). Ref = Reference streams, Con = Consolidated streams, Reh = Rehabilitated streams, and Deg = Degraded streams.*

Benthic macroinvertebrates

In general, the macroinvertebrate assemblage structure indicators did not differ among stream categories (one-way ANOVA, $p > 0.06$), except for total taxa richness ($p = 0.01$) (Table S3), which was higher in reference than in degraded streams (Fig. 3A).

Taxonomic composition of macroinvertebrate communities significantly differed between degraded streams and other stream categories (ANOSIM, $p < 0.01$; Table S3). In general, the macroinvertebrate assemblage composition indicators did not differ among stream categories (one-way ANOVA, $p > 0.07$), except for sensitive and resistant taxa richness (Table S3). Percentage sensitive taxa richness was highest in reference streams, followed by consolidated and rehabilitated streams that did not differ, and lowest in degraded streams (Fig. 3B). Percentage resistant taxa richness was lower in reference than in degraded streams (Fig. 3C).

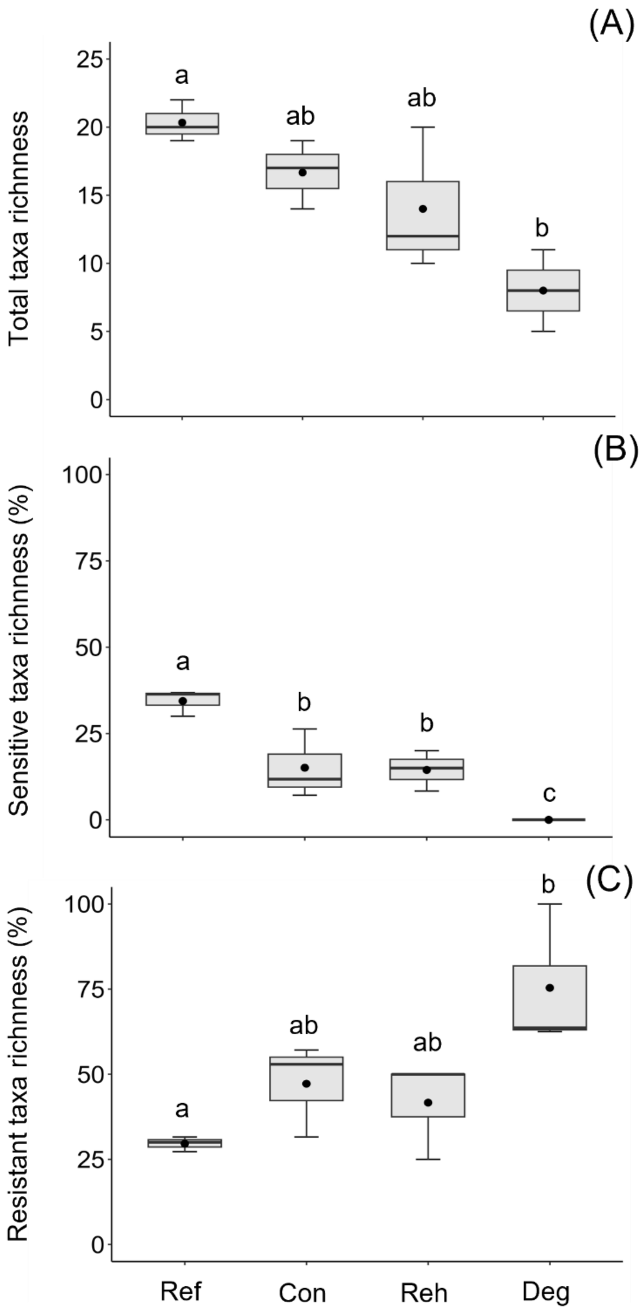


Figure 3. *Benthic macroinvertebrate assemblage structure and composition metrics: A) Total taxa richness, B) % sensitive taxa richness, and C) % resistant taxa richness. In the boxplot, the central line represents the median, the rectangle represents the 50% confidence interval, the vertical bars represent the data dispersion, and the black dot represents the mean. Different letters indicate statistical differences among stream categories (one-way ANOVA followed by Tukey's test, p < 0.05). Ref = Reference streams, Con = Consolidated streams, Reh = Rehabilitated streams, and Deg = Degraded streams.*

The biotic indices significantly differed among stream categories (one-way ANOVA, $p < 0.02$), except for the BMI ($p = 0.07$) (Table S3). In all biotic indices, except for the BMI, values were higher in reference than in degraded streams (Fig. 4). BMWP values were also higher in rehabilitated and consolidated than in degraded streams and lower in rehabilitated than in reference streams (Fig. 4A). MMI2 values were also higher for rehabilitated than degraded streams (Fig. 4D).

Strong positive correlations (Pearson's test, $r \geq 0.65$ and $p \leq 0.02$) were found between habitat diversity and total taxa richness, Shannon-Wiener, Simpson, Pielou evenness, sensitive taxa richness, tolerant individuals' abundance, BMWP, ASPT, MMI1, and MMI2 scores (Table 3). Strong negative correlations (Pearson's test, $r \leq -0.63$ and $p = 0.03$) were found between habitat diversity and resistant taxa richness, and resistant individuals' abundance (Table 3).

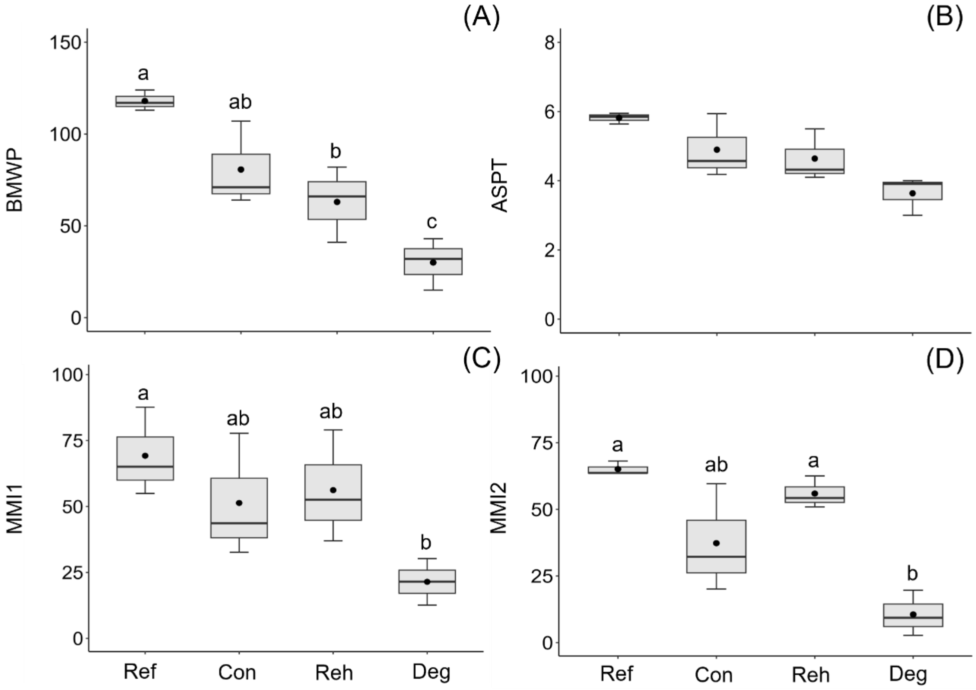


Figure 4. *Biotic indices: A) Biological Monitoring Working Party (BMWP), B) Average Score per Taxon (ASPT) by Junqueira et al. (2000, 2018), C) Macroinvertebrate Multimetric Index (MMI1) by Macedo et al. (2016), and D) Macroinvertebrate Multimetric Index (MMI2) by Silva et al. (2017). In the boxplot, the central line represents the median, the rectangle represents the 50% confidence interval, the vertical bars represent the data dispersion, and the black dot represents the mean. Different letters indicate statistical differences among stream categories (one-way ANOVA followed by Tukey's test, p < 0.05). Ref = Reference streams, Con = Consolidated streams, Reh = Rehabilitated streams, and Deg = Degraded streams.*

Table 3. *Pearson's correlations between habitat diversity and macroinvertebrate variables. The expected sign of the correlations is indicated: +, indicates a positive correlation; –, indicates a negative correlation; 0, indicates no correlation.*

Metrics	r	p-value	Expected
Total individuals' abundance	-0.27	0.39	-
Total taxa richness	**0.73**	**0.01**	+
Density of organisms per m²	-0.27	0.39	-
Shannon-Winer diversity	**0.74**	**0.01**	+
Simpson diversity	**0.65**	**0.02**	+
Pielou Evenness	**0.67**	**0.02**	+
% EPT individuals	0.47	0.12	+
% Chironomidae individuals	-0.15	0.65	-
% Chironomidae + Oligochaeta individuals	-0.47	0.12	-
EPT/Chironomidae individuals	0.55	0.06	+
EPT/(Chironomidae + Oligochaeta) individuals	0.55	0.06	+
Sensitive individuals' abundance	0.49	0.10	+
Sensitive taxa richness	**0.89**	**<0.01**	+
Tolerant individuals' abundance	**0.70**	**0.01**	-
Tolerant taxa richness	0.10	0.76	-
Resistant individuals' abundance	**-0.63**	**0.03**	-
Resistant taxa richness	**-0.64**	**0.03**	-
BMWP	**0.76**	**<0.01**	+
ASPT	**0.65**	**0.02**	+
BMI	0.52	0.08	+
MMI1	**0.68**	**0.01**	+
MMI2	**0.76**	**<0.01**	+

EPT = Ephemeroptera, Plecoptera, Trichoptera
BMWP = Biological Monitoring Working Party
ASPT = Average Score per Taxon
BMI = Benthic Multimetric Index
MMI = Macroinvertebrate Multimetric Index

Functional variables

Biofilm growth rates and Chla concentration significantly differed among all stream categories (one-way ANOVA, $p < 0.01$; Table S3), with the highest values in degraded streams followed by rehabilitated, consolidated, and reference streams (Fig. 5A and 3B). The autotrophic index significantly differed among stream categories (one-way ANOVA, $p < 0.01$; Table S3), being higher in reference and consolidated streams that did not differ among each other, than in rehabilitated and degraded streams, being higher in the former than the latter streams (Fig. 5C).

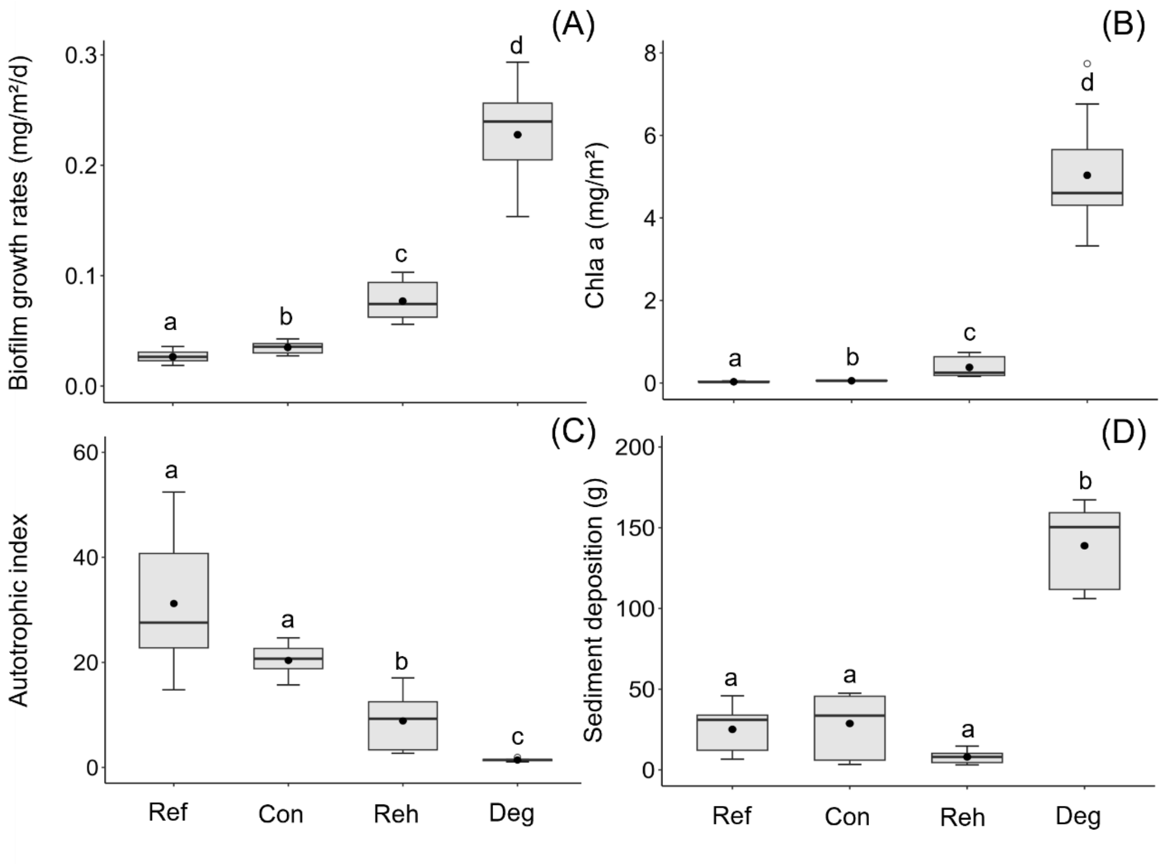


Figure 5. *Functional variables: A) Biofilm growth rate, B) Chlorophyll a concentration, C) Autotrophic index, and D) Total sediment mass deposited. In the boxplot, the central line represents the median, the rectangle represents the 50% confidence interval, the vertical bars represent the data dispersion, and the black dot represents the mean. Different letters indicate statistical differences among stream categories (one-way ANOVA followed by Tukey's test, $p < 0.05$). Ref = Reference streams, Con = Consolidated streams, Reh = Rehabilitated streams, and Deg = Degraded streams.*

Total sediment deposition significantly differed among stream categories (one-way ANOVA, $p < 0.01$; Table S3), being higher in degraded streams than in the other stream categories, which did not differ (Fig. 5D).

Percentage AFDM remaining of *B. forficata* leaf litter after 60 days incubation in the streams significantly differed among stream categories (two-way ANOVA, $p = 0.04$), but was not affected by mesh size or the interaction between factors ($p > 0.18$) (Table S3). Mass remaining was higher in degraded than in reference streams in coarse mesh bags (Fig. 6).

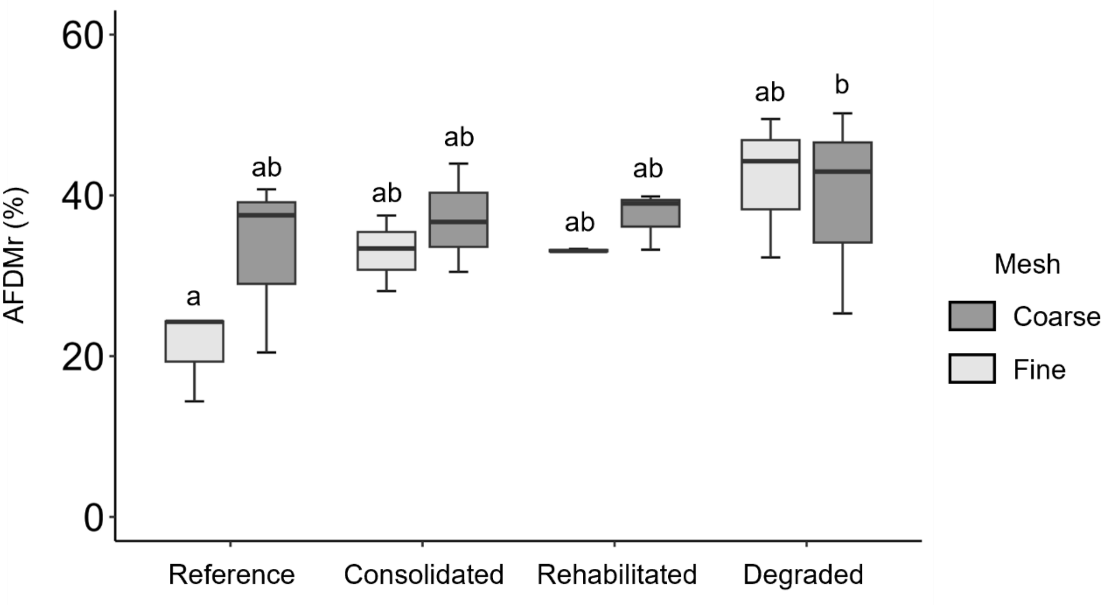


Figure 6. *Ash-free dry mass remaining (AFDMr) in coarse and fine mesh of* Bauhinia forficata *leaf litter after 60 days of stream incubation. In the boxplot, the central line represents the median, the rectangle represents the 50% confidence interval, the vertical bars represent the data dispersion, and the black dot represents the mean. Different letters indicate statistical differences among stream categories (two-way ANOVA followed by Tukey's test, p < 0.05).*

Discussion

We confirmed the hypothesis that there would be a gradient of structural and functional integrity, going from the reference to the degraded habitat condition. We found significant improvements in habitat diversity, primary production, sediment deposition and percentage of sensitive benthic macroinvertebrate taxa in rehabilitated compared with degraded streams. These results show the effectiveness of rehabilitation actions in improving the ecological condition of streams, moving them away from the state of environmental degradation and placing them at the same level as moderately degraded streams (consolidated). However, rehabilitated streams are not at the same level as reference sites, indicating that there was no "recovery" to natural conditions, which corroborates the idea that the return to the natural state in an urbanized area is an unrealistic a short-term goal (Hughes et al., 2014; da Silva & Porto, 2021).

Rehabilitation improves habitat structure

Considering the physical and chemical parameters of the water, it was expected that there would be a difference between the categories of streams (Ramirez et al., 2014; Macedo et al., 2022). However, few parameters differed and, in general, only between reference and degraded streams. The absence of significant differences among streams categories may be due to the reduction in environmental heterogeneity caused by urbanization. The indirect effects of urbanization (e.g., surface runoff, groundwater contamination, human use for recreation) can act as a homogenizer of the stream basin, even in reference streams (Fausch et al., 2002, Wiens, 2002; Paul & Meyer, 2008; Loflen et al., 2016). Within a stream basin, homogeneous environments offer a smaller range of conditions (e.g.,. water pH, flow velocity). In addition, there may also be an effect of the great variability within stream categories, which makes it difficult to identify water quality standards.

We was expected to find a gradient in habitat diversity going from the reference to the degraded. Our results showed significant differences among reference, rehabilitated and degraded streams, but there were no differences between the rehabilitated and consolidated streams, confirming our prediction. This assessment provides important information on physical integrity and the influence of local and regional anthropogenic disturbances (Callisto et al., 2002; Kieling-Rubio et al., 2015; Larentis, et al., 2021) and shows that the interventions carried out resulted in improving stream condition that was previously degraded to moderately altered (similar to the consolidated ones). However, they are still distinct from the "close to natural" or reference state. This may reflect the fact that rehabilitation works, in general, focus on improvements in water quality, aesthetics and the use of the space as a park open to public visitation and not on the full recovery of ecosystem (Paul & Meyer, 2008; Wantzen et al., 2019). In contexts as altered as the urban one, the full recovery of the ecosystem is generally no longer achievable because land use has changed and it is not possible revert urban occupation to forest occupation. In addition, studies show that the "recovery" of degraded areas to their natural state is almost unlikely, being an unrealistic goal (Hughes et al., 2014; da Silva & Porto, 2021).

Habitat diversity was significantly correlated with taxa richness, diversity, assemblage composition and multimetric indices of benthic macroinvertebrates. This indicate that improvements in habitat diversity, physical stream structure and the riparian zone can lead to positive changes in benthic macroinvertebrate assemblages. This is because, in addition to the absence of environmental pollution, factors such as organic matter input (e.g., leaves, flowers, fruits and wood) (Linares et al., 2021), channel bottom composition (Heino et al., 2018) and flow variations (Bouckaert & Davis, 1998; Calderon et al., 2023) are important factors that influence the presence/absence of organisms and, therefore, greater environmental heterogeneity may imply coexistence of more organisms with different preferences (McCreadie & Bedwell, 2013; Agra et al., 2021).

To improve the classification of rehabilitated streams, measure such as increasing the width and diversity of riparian vegetation cover, for example, could lead to further improving habitat diversity (all streams obtained a minimum score in these parameters; Table S4) (Pedraza et al., 2018; Fonseca et al., 2021; Linares et al., 2021).

Rehabilitation improves macroinvertebrate assemblage structure and composition

That the assemblages of benthic macroinvertebrates would show better structure and composition in the reference streams, followed by the consolidated and rehabilitated streams compared to the degraded streams, was weakly supported. It was expected that rehabilitation would promote improvement in environmental quality, differentiating them from degraded conditions (Al-Zakana et al., 2020; Zerega et al., 2021). However, biological rehabilitation occurred to a lesser extent than expected. Although the rehabilitated streams, in general, did not differ from the reference and consolidated streams, they also did not differ from the degraded streams, except for sensitive taxa richness, BMWP and MMI2, indicating that they are in an intermediate situation between degradation and reference.

Previous studies by Palmer et al. (2014) and Kail et al. (2015) showed that biological differences between environmental conservation categories are not always detected by richness and diversity indices, but rather by relative abundances or taxonomic composition. Here, for the composition of the benthic communities, differences were only detected between the rehabilitated and degraded streams for the percentage of sensitive taxa richness. The increase in percentage of sensitive taxa richness is related to the improvement of environmental quality (Feio et al., 2015; Sterling et al., 2016). This result shows the importance of rehabilitation in recovering animal biodiversity, even if by litlle. Similar results were found in other studies of urban regeneration projects in countries in the Global North, which showed mixed biological outcomes, with only 5% – 20% showing significant biological improvements (Al Zankana et al., 2020).

Of all biological indices tested, only BMWP and MMI2 were able to detect differences between rehabilitated and degraded streams. These two indices, which quantify and combine several measures into a single indicator, if used in conjunction with other assessment and forecasting tools for freshwaters, can provide useful and easily communicated information as a basis for protecting and rehabilitating degraded environments (Statzner & Bêche, 2010; Martins et al., 2022; Vadas et al., 2022).

The lack of differences between rehabilitated and reference and degraded streams may be due to the high variability found within stream categories, suggesting the need to better categorize the streams into functional types. This likely reduced our ability to detect changes by underestimating the true effect size associated with the category. In addition, it may be linked to the legacy effect of anthropic disturbances that reverberate for a long time in communities, making their resilience difficult, even in the reference parks, which, despite being in the best possible conditions, still suffer urban pressures (Allan, 2004; Camana et al., 2020; Linares et al., 2022). In addition, the streams are not connected with other nearby streams in a high state of conservation that could act as potential colonizers nearby. In this way, the streams may even have improved their physical condition, but the absence of a source of colonizers (streams in good condition that can provide organisms to colonize the rehabilitated stream) it may not be able to recover the biological communities (Korsu, 2004; Zerega et al., 2021).

Rehabilitation improves ecosystem functioning

As expected found large differences between stream categories for chlorophyll a concentration, biofilm growth rate on artificial substrates, and the autotrophic index. Previous studies found a relationship between increased periphyton biomass and increased nutrient input (Bourassa & Cattaneo, 2000, Feio et al., 2010). In addition, decreases in channel shading often leads to increases in chlorophyll a concentration (Reisinger et al., 2019), which allows the biofilm to have a greater nutrient absorption capacity (Burrell et al., 2014). We found the highest values of chlorophyll a and biomass in degraded streams, followed by rehabilitated and consolidated streams and the lowest values in reference streams, likely due to higher availability of dissolved nitrogen (nutrient enrichment) and reduced vegetation cover, with increased substrate exposure to light, in degraded streams.

We expected to find less organic matter mass remaining (i.e., faster decomposition) in the reference streams, followed by consolidated and rehabilitated and more mass reining in the degraded streams. Decomposition of organic matter is generally promoted by moderate nutrient enrichment (Woodward et al., 2012; Rosemond et al., 2015), presence of microbial decomposers (Gulis et al., 2019) and invertebrate shredders (Ferreira et al., 2006) and can be inhibited by water acidification (Dangles et al., 2004; Ferreira & Guérold, 2017). Our results showed significant differences in mass remaining between reference and degraded streams. This may be associated with the reduced presence of shredders, which include taxa that are very sensitive to environmental disturbances (e.g., Plecoptera and Trichoptera), such as nutrient load, metal contaminants and changes in streams channels (Dangles et al., 2004; Piscart et al., 2009), in degraded streams. However, we did not detect differences between rehabilitated or consolidated stream and other stream categories. Organic matter decomposition may not show clear responses in moderately impacted environments if other confounding factors are at play (Ferreira et al., 2020), and identical decomposition rates can be observed at very different levels of nutrient loading (Woodward et al., 2012).

For the sediment deposition we expected to find a significant increase in degraded streams, followed by rehabilitated, consolidated streams and little sediment in reference streams. Previous studies have found a relationship between land use and deposition and the characteristic of the deposited sediment (Rosi-Marshall et al., 2016; dos Reis Oliveira et al., 2018, 2020). Interestingly, the rehabilitated streams were those that showed the lowest sediment deposition, probably due to the containment and stabilization works on the banks implemented in the streams and, therefore, retention of the sediment that would be transported into the streams (Mount et al., 2002; Florsheim & Mount, 2003).

Conclusion

Our results showed that the rehabilitated streams, after 15 years of the intervention, are in better structural and functional conditions than the degraded streams, without strongly differing from the streams in a moderately altered condition (consolidated), but they did not reach a condition similar to that of the reference streams. We conclude that rehabilitation is effective in removing the site from the state of degradation, improving the structure and functioning of the ecosystem. Furthermore, the combined use of functional and structural indicators allowed an integrated assessment of the ecological integrity of the stream and allowed the detection of differences between streams categories beyond those detected based on single water quality indicators.

This study contributes data and critical multi-tool information about the relevance of environmental rehabilitation of urban streams in the third most populous Brazilian metropolis. We hope these results support more appropriate options for managing urban streams in tropical regions.

Future perspective's

Future studies should explore: (1) We could assess the three streams separately to determine possible differences in degrees of recovery. (2) Rehabilitation effects on streams in Brazilian urban areas vary seasonally (dry season versus wet season). (3) The use of Odonata's nymphal and adult stages as bioindicators to better understand the effects of urbanization on biodiversity.

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Author Contributions

K.H.M. planned the study, performed field sampling, sample processing, data analysis and writing. V.F. and M.C. contributed to plan this study, data analysis and writing. All authors have read and approved the final manuscript.

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Conflict of interests

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Supplementary material

The way from degradation to reference conditions: rehabilitation of urban streams improves their structure and functioning

Karoline H. Madureira[a]*, Verónica Ferreira[b] & Marcos Callisto[a]

[a]Laboratório de Ecologia de Bentos, Departamento de Genética, Ecologia e Evolução, Instituto de Ciências Biológicas, Universidade Federal de Minas Gerais, Avenida Antônio Carlos, 31270-901 Belo Horizonte, Brazil

[b]MARE – Marine and Environmental Sciences Centre, ARNET – Aquatic Research Network, Department of Life Sciences, University of Coimbra, Calçada Martim de Freitas, 3000–456 Coimbra, Portugal

E-mail addresses: karolhmadureira@gmail.com, veronica@ci.uc.pt, mcallisto13@gmail.com

*Corresponding author: karolhmadureira@gmail.com

Table S1. *Location of the study streams in Minas Gerais, Brazil.*

Stream categories	Park	Stream	Hydrographic basin	Geographic coordinates	Elevation (m)
Reference	Serra do Rola Moça State Park	Taboões	Bacia do Ribeirão Arrudas	20°03'39''S 44°03'01''W	1127
Reference	Mangabeiras Municipal Park	Riacho da Serra	Bacia do Ribeirão Arrudas	19°56'43''S 43°54'22''W	1068
Reference	Roberto Burle Marx Municipal Park	Clemente	Bacia do Ribeirão Arrudas	20°00'02''S 43°59'47''W	1019
Consolidated	Jacques Custeau Municipal Park	Riacho do Bonsucesso	Bacia do Ribeirão Arrudas	19°58'20''S 43°59'10''W	974
Consolidated	Fazenda Lagoa do Nado Municipal Park	Riacho do Nado	Bacia do Ribeirão Isidoro	19°50'28''S 43°57'25''W	800
Consolidated	Aggeo Pio Sobrinho Municipal Park	Riacho da Ponte Queimada	Bacia do Ribeirão Arrudas	19°58'41''S 43°58'17''W	935
Rehabilitated	Primeiro de Maio Ecological Park	Primeiro de Maio	Bacia do Ribeirão do Onça	19°51'16''S 43°55'48''W	810
Rehabilitated	José Lopes dos Reis Baleares Municipal Park	Baleares	Bacia do Ribeirão Isidoro	19°48'07''S 43°57'54''W	803
Rehabilitated	Nossa Senhora da Piedade Municipal Park	Sem nome	Bacia do Ribeirão do Onça	19°50'51''S 43°55'38''W	812
Degraded	Technologic Park of Belo Horizonte	Riacho do Mergulhão	Bacia do Ribeirão do Onça	19°53'03''S 43°58'37''W	859
Degraded	Zoobotanical Garden of Belo Horizonte	Bom Jesus	Bacia do Ribeirão do Onça	19°51'43''S 44°01'04''W	827
Degraded	Vilarinho Municipal Park	Lagoinha	Bacia do Ribeirão Isidoro	19°48'05''S 43°59'2''W	779

Table S2. *Macroinvertebrates collected in 12 study streams. BL = Bank of leaves. ST/GR =Stone or gravel. Sa = Sand or fine sediment*

	Reference									Consolidated								
	TB			MB			BM			JC			LN			AP		
Taxa	**BL**	**St/Gr**	**Sa**	**BL**	**St/Gr**	**Sa**	**BL**	**St/Gr**	**Sa**	**BL**	**St/Gr**	**Sa**	**BL**	**St/Gr**	**Sa**	**BL**	**St/Gr**	**Sa**
Insecta																		
Ephemeroptera																		
Baetidae	6			3	1	1	17	7		14						9	10	1
Leptohyphidae	12	1	10				6	17								2		
Leptophlebiidae	24	1	2		7		14	61								9		
Plecoptera																		
Gripopterygidae	1							3										
Perlidae	1			2			13	4								1		
Trichoptera																		
Calamoceratidae	1			32	1	3	5						2			4		
Hydroptilidae	12	1			1		6	12		10								
Hydropsychidae	16	5					18	16	1	42	1					17	4	2
Leptoceridae				47														
Polycentropodidae				1	2		1	1										
Odonata																		
Calopterigidae	10	2	1										1			1		
Coenagrionidae																	1	
Corduliidae					1		2					3	1			2	2	1
Dicteriadidae																3		
Gomphidae									1									
Libellulidae												1					3	2
Megapodagrionidae				1										1				
Perilestidae				1	6													
Heteroptera																		
Gelastocoridae												1						
Mesoveliidae											5							
Naucoridae	3																	
Notonectidae				1														
Veliidae									2			3						7
Coleoptera																		
Dytiscidae				9														
Elmidae	11			62	7	5	8	31	2				1		1	3		
Hydrophilidae							2						4	1	2			
Lutrochidae			19	8														

Megaloptera																		
Corydalidae	1																	
Neuroptera							2											
Lepidoptera																		
Pyralidae							1											
Diptera																		
Ceratopogonidae	1					9	3		3	2		4	1	1				
Chaoboridae																		
Chironomidae	60	6	25	629	143	92	99	74	44	816	62	100	90	28	16	128	66	28
Dixidae																1		
Empididae	1	1	2	2	1		1			1		1	1					
Muscidae												1						
Simuliidae	37	1						7		30	4						4	
Tabanidae												1						
Tipulidae				1				1				1				2		
Annelida																		
Oligochaeta	1			2	1	2	1	2	1		1		2					
Rhynchobdellida																		
Glossiphoniidae										2				1				
Gastropoda																		
Ampularidae																		
Ancilidae													21	8	3	14		
Hydrobiidae													2	2				
Lymnaeidae																		
Physidae																		
Planorbidae																		
Thiaridae														1	4			
Bivalvia				28														1
Acari																		
Hydracarina	1																	
Richness	18	8	6	16	11	6	17	13	7	8	5	10	11	8	5	14	7	7
Abundance	199	18	59	829	171	112	199	236	54	917	73	116	126	43	26	196	90	42
Richness/stream		19			20			22			17			14			19	
Abundance/stream		276			1112			489			1106			195			328	
Richness/stream type					20									17				
Abundance/stream type					626									543				

	Rehabilitated									Degraded								
	PM			BL			NS			BT			ZO			VI		
Taxa	**BL**	**St/Gr**	**Sa**	**BL**	**St/Gr**	**Sa**	**BL**	**St/Gr**	**Sa**	**BL**	**St/Gr**	**Sa**	**BL**	**St/Gr**	**Sa**	**BL**	**St/Gr**	**Sa**
Insecta																		
Ephemeroptera																		
Baetidae			1		1		5	2	1				80	27				
Leptohyphidae	1		1				11											
Leptophlebiidae																		
Plecoptera																		
Gripopterygidae																		
Perlidae																		
Trichoptera																		
Calamoceratidae					3													
Hydroptilidae							25	8										
Hydropsychidae	1	1	2				390	21	9									
Leptoceridae																		
Polycentropodidae																		
Odonata																		
Calopterigidae					3		8											
Coenagrionidae				1		1												
Corduliidae	3			3	8	2											1	
Dicteriadidae																		
Gomphidae																		
Libellulidae				1			1								1			
Megapodagrionidae																		
Perilestidae																		
Heteroptera																		
Gelastocoridae																		
Mesoveliidae							1											
Naucoridae																		
Notonectidae																		
Veliidae																		
Coleoptera																		
Dytiscidae				2													1	
Elmidae						2	5		1				2	3	2	1		
Hydrophilidae	7	3	2	4	6	2	2		1				7					
Lutrochidae																		

Megaloptera																		
Corydalidae																		
Neuroptera																		
Lepidoptera																		
Pyralidae							1											
Diptera																		
Ceratopogonidae						15							1		1	1	1	1
Chaoboridae							1											
Chironomidae	21	120	11	50	5	9	487	54	45	2			528	204	289	185	23	88
Dixidae																		
Empididae													3					
Muscidae																		
Simuliidae								1					20	40	1			
Tabanidae																		
Tipulidae												1						
Annelida																		
Oligochaeta							2		5	41	734	1202	22	4	30	1	3	1
Rhynchobdellida																		
Glossiphoniidae	2						3		2				56					
Gastropoda																		
Ampularidae																1	1	1
Ancilidae			1				13		2									
Hydrobiidae																		
Lymnaeidae		1					3			35								
Physidae		1	2	143	13	23	17	1		203	22	12	2			3		
Planorbidae							1											
Thiaridae							2											
Bivalvia																		
Acari																		
Hydracarina																		
Richness	6	5	7	7	7	7	19	6	8	4	2	3	10	5	6	6	6	4
Abundance	35	126	20	204	39	54	978	87	66	281	756	1215	721	278	324	192	30	91
Richness/stream		10			12			20			5			11			8	
Abundance/stream		181			297			1131			2252			1323			313	
Richness/stream type					14									8				
Abundance/stream type					536									1296				

Table S3. *Summary table for one-way ANOVAs (most variables; factor: stream category), two-way ANOVA (for AFDM remaining; factors: stream category and mesh) and ANOSIM (for taxonomic composition).*

Metric	df	F	p-value
Abiotic characterization			
Conductivity	3	19.38	0.00
Dissolved oxygen concentration	3	4.63	0.04
Dissolved oxygen saturation	3	4.29	0.04
Oxygen biochemical demand	3	14.4	0.00
pH	3	0.32	0.81
Redox potential	3	1.99	0.19
Resistivity[1]	3	10.58	0.00
Temperature	3	1.42	0.31
Total solid dissolved	3	15.87	0.00
Total nitrogen	3	0.24	0.87
Total phosphorus[1]	3	0.58	0.65
Turbidity	3	1.02	0.43
Canopy cover[1]	3	8.31	0.01
Depth	3	0.56	0.66
Flow velocity	3	3.69	0.06
Wet width	3	1.97	0.20
Habitat diversity	3	39.23	0.00
Functional variables			
Biofilm growth rates[1]	3	269.6	0.00
Chlorophyll a[1]	3	390	0.00
Autotrophic index	3	91.95	0.00
Sediment deposition	3	145	0.00
AFDM remaining			
Stream category	3	3.26	0.04
Mesh	1	1.97	0.18
Interaction	3	0.88	0.47
Benthic macroinvertebrate			
Structure			
Total individuals abundance	3	0.98	0.45
Total taxa richness	3	7.1	0.01
Density	3	0.98	0.45
Shannon H' diversity index	3	3.68	0.06
SimpsonH' diversity index[1]	3	2.53	0.13
Pielou evenness index	3	2.21	0.17
Composition			
Taxonomical composition	-	-	0.01
% EPT individuals	3	1.61	0.26
% Chironomidae individuals	3	0.34	0.80
% Chironomidae + Oligochaeta individuals	3	2.25	0.16
EPT/Chironomidae individuals	3	2.24	0.16
EPT/Chironomidae + Oligochaeta individuals	3	2.23	0.16
Sensitive individuals abundance	3	1.98	0.20

Sensitive taxa richness	3	16.04	0.00
Tolerant individuals abundance	3	4.93	0.03
Tolerant taxa richness	3	0.88	0.49
Resistant individuals abundance	3	3.59	0.07
Resistant taxa richness	3	5.27	0.03
Biotic indices			
Biological Monitoring Working Party (BMWP)	3	13.56	0.00
Average Score Per Taxa (ASPT)	3	5.49	0.02
Benthic Multimetric index (BMI)	3	3.44	0.07
Macroinvertebrate Multimetric Index 1 (MMI1)	3	5.76	0.02
Macroinvertebrate Multimetric Index 2 (MMI2)	3	13.25	0.00

[1]Log(x)-transformed

EPT = Ephemeroptera + Plecoptera + Trichoptera

***Table S4. Benthic macroinvertebrate metrics** per stream; values are means (n=3).*

Stream category	Reference			Consolidated			Rehabilitated			Degraded		
Stream	Rola Moça	Mangabeiras	Burle Marx	Jacques Custeau	Lagoa do Nado	Aggeo Pio	Primeiro de Maio	Baleares	N. Sra Piedade	BhTec	Zoológico	Vilarinho
Abundance	276	1112	489	1106	195	328	181	297	1131	2252	1323	313
Richnness	19	20	22	17	14	19	10	12	20	5	11	8
Density	3066.67	12355.56	5433.33	12288.89	2166.67	3644.44	2011.11	3300.00	12566.67	25022.22	14700.00	3477.78
Shannon	2.21	1.02	1.97	0.59	1.17	1.41	0.73	1.29	1.21	0.43	0.91	0.31
Simpson	0.84	0.39	0.76	0.22	0.50	0.53	0.29	0.58	0.59	0.22	0.39	0.11
Evenness	0.75	0.34	0.64	0.21	0.44	0.48	0.32	0.52	0.41	0.26	0.38	0.15
EPT richness	8	7	9	3	1	6	3	2	4	0	1	0
EPT/Chironomidae individuals	1.02	0.12	0.93	0.07	0.01	0.27	0.05	0.06	0.81	0.00	0.10	0.00
EPT/(Chironomidae + Oligochaeta) individuals	1.01	0.12	0.91	0.07	0.01	0.27	0.05	0.06	0.80	0.00	0.10	0.00
% EPT individuals	33.70	9.08	41.31	6.06	1.03	17.99	3.87	1.35	41.73	0.00	8.09	0.00
% Chironomidae individuals	32.97	77.70	44.38	88.43	68.72	67.68	83.98	21.55	51.81	0.09	77.17	94.57
% Chironomidae + Oligochaeta individuals	33.33	78.15	45.19	88.52	69.74	67.68	83.98	21.55	52.43	87.88	81.41	96.17
% Sensitive taxa richnness	31.52	8.63	36.40	4.79	1.03	11.89	3.31	1.01	41.03	0.00	0.00	0.00
% Sensitive taxa abundance	36.84	30.00	36.36	11.76	7.14	26.32	20.00	8.33	15.00	0.00	0.00	0.00
% Tolerant taxa richnness	19.20	9.53	15.34	2.44	6.15	13.72	8.84	12.12	2.48	0.00	9.22	0.96

% Tolerant taxa abundance	31.58	40.00	36.36	35.29	35.71	42.11	30.00	66.67	35.00	0.00	36.36	37.50
% Resistant taxa richnness	49.28	81.83	48.26	92.77	92.82	74.39	87.85	86.87	56.50	100.00	90.78	99.04
% Resistant taxa abundance	31.58	30.00	27.27	52.94	57.14	31.58	50.00	25.00	50.00	100.00	63.64	62.50
BMWP	112	113	138	79	71	123	50	69	102	15	45	36
ASPT	5.9	5.7	6.3	4.6	5.1	6.5	5.0	5.8	5.1	3.0	4.1	4.5
BMI	30	24	28	22	24	28	22	26	30	6	24	16
MMI1	87.68	52.57	66.01	33.33	44.35	78.18	54.97	78.51	38.89	11.79	29.86	22.35
MMI2	63.68	63.23	68.22	18.65	32.56	59.02	62.59	51.81	49.91	9.33	16.53	2.76

EPT, Ephemeroptera+Plecoptera+Trichoptera; BMWP, Biological Monitoring Working Party; ASPT, Average Score Per Taxon; BMI, Benthic Multimetric Index; MMI1, Macroinvertebrate Multimetric Index 1; MMI2, Macroinvertebrate Multimetric Index 2

Table S5. *Local physical and chemical parameters per stream; values are means (n = 5).*

Stream category	Reference			Consolidated			Rehabilitated			Degraded		
Stream	Taboões	Mangabeiras	Burle Marx	Jacques Custeau	Lagoa do Nado	Aggeo Pio	Primeiro de Maio	Baleares	Nossa Senhora	BhTec	Zoológico	Vilarinho
Temperature (°C)	20.4	17.8	18.4	18.7	21.1	17.9	21.5	19.0	22.8	20.2	21.0	19.6
pH	7.90	7.33	7.78	7.56	7.49	7.65	7.39	7.95	8.03	7.96	7.37	7.72
Turbidity (NTU)	0.2	2.0	0.8	5.0	4.0	0.8	2.2	0.6	7.0	1.8	3.6	20.8
Conductivity (μS/cm)	11.7	70.7	26.2	78.5	211.5	68.5	320.9	407.4	285.8	374.0	277.6	304.3
Total solids dissolved (ppm)	6.8	43.5	22.1	48.0	122.3	47.6	191.4	273.2	166.5	223.3	164.2	186.6
Resistivity (Ωm)	84.2	13.5	25.1	11.7	7.0	12.9	5.1	2.4	4.0	3.0	4.7	3.3
Oxi-redox potential (mV)	175.8	142.7	141.7	116.6	103.4	122.4	120.9	172.6	121.1	147.8	126.7	116.6
Dissolved oxygen (%)	105.0	99.8	99.8	93.0	77.2	96.6	87.6	89.8	85.4	89.6	42.4	51.0
Dissolved oxygen (mg/L)	8.4	8.3	8.4	7.9	6.3	8.1	7.2	7.6	6.8	7.4	3.5	4.2
Current velocity (m/s)	0.5	0.1	0.4	0.1	0.0	0.0	0.0	0.1	0.0	0.8	1.0	0.1
Width (m)	2.59	3.53	1.49	2.99	4.09	2.87	1.82	2.04	2.32	2.73	2.15	2.67
Canopy cover (%)	85.6	80.7	84.8	78.3	78.5	79.0	68.3	67.6	75.3	76.2	68.3	1.0
Depth (m)	0.15	0.13	0.11	0.10	0.08	0.07	0.07	0.07	0.08	0.08	0.07	0.06
Oxygen biochemical demand (mg/L)	0.5	0.4	0.7	1.2	0.8	0.7	1.1	0.7	1.8	2.0	3.1	3.1
Total nitrogen (mg/L)	5.04	7.82	3.95	8.61	7.90	3.80	5.40	4.45	9.32	13.49	7.75	2.55
Total phosphorus (mg/L)	0.164	0.045	0.081	0.103	0.023	0.014	0.019	0.145	0.018	0.105	0.060	0.017

Table S6. *Functional variables per stream; values are means (n =5).*

Stream category	Reference streams			Consolidated streams			Rehabilitated streams			Degraded streams		
Stream	Taboões	Mangabeiras	Burle Marx	Jacques Custeau	Lagoa do Nado	Aggeo Pio	Primeiro de Maio	Baleares	Nossa Senhora	BhTec	Zoológico	Vilarinho
Biofilm growth (mg/m2/d)	0.027	0.027	0.026	0.035	0.035	0.036	0.090	0.067	0.074	0.194	0.224	0.277
Chlorophyll a (mg/m^2)	0.020	0.045	0.024	0.055	0.058	0.050	0.238	0.691	0.207	3.970	4.651	6.961
Autotrophic index	41.511	18.131	33.968	19.514	19.173	22.444	12.165	3.109	11.280	1.509	1.472	1.209
Total sediment deposition (g)	35.257	9.239	30.740	36.141	4.760	45.258	9.018	3.841	11.093	110.292	161.957	146.039
AFDMr (%) fine mesh	20.455	40.754	37.526	43.948	30.483	36.697	38.985	33.242	39.866	50.207	25.313	42.949
AFDMr (%) coarse mesh	14.375	24.380	24.267	37.496	28.096	33.390	33.330	32.989	33.085	49.498	32.278	44.252

AFDMr, Ash-free dry mass remaining

CONSIDERAÇÕES FINAIS

Essa dissertação buscou entender o estado de integridade ecológica de riachos urbanos reabilitados. Para tal, indicadores estruturais (parâmetros físicos e químicos de coluna d'água, diversidade de habitat e integridade de zona ripária e macroinvertebrados bentônicos) e funcionais (produção primária, decomposição de detritos foliares e deposição de sedimentos) foram utilizados para avaliar a qualidade ecológica. Vimos que riachos reabilitados, em geral, encontram-se em melhor integridade ecológica que riachos degradados, se aproximando dos riachos consolidados. No entanto, os riachos reabilitados não estão próximos às condições encontradas nos riachos em condições de referência. Concluímos que a reabilitação é eficaz em retirar o local do estado de degradação, melhorando a estrutura e o funcionamento do ecossistema. No entanto, a recuperação completa do estado ecológico de riachos urbanos é possivelmente irrealista. Além disso, a utilização conjunta de indicadores estruturais e funcionais permitiu avaliar a integridade ecológica de riachos urbanos, identificando diferenças entre categorias de riachos além daquelas detectadas com base em indicadores únicos de qualidade da água.

Dessa forma, esse trabalho contribuiu para preencher lacunas de conhecimento referentes aos riachos reabilitados em regiões tropicais e como avaliar a integridade ecológica de riachos urbanos.

PERPECTIVAS FUTURAS

Para o desenvolvimento de estudos futuros recomendo:

- Avaliar os três riachos separadamente para determinar possíveis diferenças nos graus de recuperação.
- Determinar se os efeitos da reabilitação em áreas urbanas brasileiras variam sazonalmente (estação seca versus estação chuvosa).
- Avaliar diferenças na biomassa e produção secundária da comunidade de macroinvertebrados bentônicos.
- Usar Odonata em fase de ninfa e adultos como bioindicadores para melhor compreender os efeitos da urbanização sob à biodiversidade.

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